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FORWARD

The purpose of the National Workshop on Biological Monitoring and Criteria was to evaluate and articulate the role of instream biosurvey data in the USEPA and State surface water quality programs. The Workshop participants were from various State and Federal agencies throughout the country and provided the needed perspectives to ensure that biological survey data will be used to support the goals, objectives, and policies of the Clean Water Act. The Proceedings provide a special forum, separate from the National Biological Monitoring and Criteria Workshop Report, in which invited speakers were asked to address recent biological criteria developments. We thank each of the authors for their efforts.

The Region V, Instream Biocriteria and Ecological Assessment Committee wishes to thank Deborah White, La Vennecy Brown, Charles Steiner, Max Anderson, and Don Krichiver for their technical assistance and professional attitude in ensuring that these proceedings were completed. Special thanks also to managers which facilitated the IBEAC editorial committees duties by changing schedules and allowing adequate time for completion of the papers: James Luey, James Giattina, Kenneth Fenner, Charles Sutfin, Noel Kohl, Curtis Ross, William Sanders III, Frank Thomas, and Andrea Jirka.

The impetus for this Workshop was provided by recent amendments in the Clean Water Act. The Clean Water Act policy "that the discharge of toxic pollutants in toxic amounts be prohibited" (section 101[a][3]) supports the "national goal that the discharge of pollutants into the navigable waters be eliminated" (section 101[a][1]). The intent of the commonly cited "no toxic discharge in toxic amounts" is to protect instream biological communities, wild and domestic animal life, and human health.

In 1984, USEPA published the national "Policy for the Development of Water Quality-Based Permit Limitations for Toxic Pollutants" (FR 49 [48]: 9016-9019, March 9). The "Statement of Policy" included the requirements that "states will use biological techniques and available data on chemical effects to assess toxicity impacts and human health hazards based on the general standard of 'no toxic materials in toxic amounts'", and "under section 308 and section 402 of the Clean Water Act (the Act), EPA or the State may require NPDES permit applicants to provide chemical, toxicity, and instream biological data necessary to assure compliance with standards." It continues by stating "where there is a significant likelihood of toxic effects to biota in the receiving water, EPA and the States may impose permit limits on effluent toxicity and may require an NPDES permittee to conduct a toxicity reduction evaluation."

The Water Quality Act of 1987 requirements of Sections 308(c) and 308(d), amend the Clean Water Act as follows:

Section 308(c) of the Water Quality Act of 1987 states that "Section 304(a) is amended by adding [that]...after consultation with appropriate

State agencies and within 2 years after the date of enactment of the Water Quality Act of 1987, [EPA] shall develop and publish information on methods for establishing and measuring water quality criteria for toxic pollutants on other bases than pollutant-by-pollutant criteria, including biological monitoring and assessment methods."

In addition, Section 308(d) of the Water Quality Act of 1987 states that "Section 303(c)(2) is amended by inserting...Where such numerical criteria [for toxics] are not available, whenever a State reviews water quality standards pursuant to paragraph (1), or revises or adopts new standards pursuant to this paragraph, such State shall adopt criteria based on biological monitoring or assessment methods consistent with information published pursuant to Section 304(a)(8)."

The preceding sections of the Clean Water Act clearly identify the need for establishing and supporting water quality assessments and criteria based upon direct measurement of the indigenous aquatic communities. As a result of the Workshop, and a Region V Policy Statement presented at the Workshop, USEPA is developing a National Biocriteria Policy and National Biocriteria Guidance to support the Water Quality Act of 1987 Sections 308(c) and 308(d). USEPA is supporting those States using instream biological survey data to establish and measure water quality criteria for toxic and conventional pollutants, and is encouraging those States not active in this process to develop program plans for implementation.

Wayne S. Davis,
Local Conference Host
Chairperson, Instream Biocriteria and Ecological
Assessment Committee

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SURFACE WATER MONITORING

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Introduction

Surface Water Monitoring

After focusing on technology-based water pollution controls for well over a decade, Federal and State agencies are shifting the emphasis to water quality-based approaches for solving the remaining (post-BAT) problems. As highlighted in the 1987 Water Quality Act (WQA 1987), assessments of ambient conditions (e.g., Sections 305(b), 304(1), 314 and 319) should play an important role in implementing these approaches. Ambient data are needed to identify problem waterbodies, set management priorities, develop water quality-based controls, and document the effectiveness of these controls. However, as pointed out in EPA's recent report, "Surface Water Monitoring a Framework for Change" (USEPA 1987), it is unlikely that existing monitoring programs will be able to fulfill these data needs.

Changing Needs

There are solid facts supporting this prediction. One is that the list of potentially important pollutants has expanded tremendously in the last decade. Many of these pollutants are toxic substances that can cause deleterious effects at levels that are very difficult to detect in the ambient environment. Others, e.g. fine sediment loadings and habitat loss, defy traditional toxicological characterization and measurement. Furthermore, the impact of these stress agents is not simply dependent upon exposure concentration. Duration and frequency of exposures and the influence of site-specific water quality factors are also important. These factors interact in a continually varying environment to profoundly influence the actual expression of effect. It is this need to characterize the actual as well as the predicted effect of pollutants that poses the greatest challenge to existing monitoring programs.

Traditional programs have focused almost entirely on analysis of water column chemistry using a mix of fixed station and intensive survey monitoring. Fixed stations supposedly provide the broad geographical coverage needed to screen for emerging water quality problems and characterize general trends. Intensive surveys, on the other hand, supply the more detailed information needed to diagnose the causes of specific problems and develop appropriate controls. And when conventional pollutants (e.g., BOD, TSS, pH) emanating from point sources are the principle concern, these programs can work quite well. However, the bewildering array of pollutants and their complex chemical behavior instream, coupled with the sheer expense of analyzing for them, makes routine monitoring for all but a few of them infeasible. Therefore, analytical resources must be effectively targeted on waterbodies where <u>real</u> problems, not merely predicted problems, actually exist.

Bioassessments

Bioassessments can help. Such assessments measure the direct responses of instream organisms exposed to environmental pollutants rather than just the exposures. The rationale behind this approach is that the resident organisms (communities, populations, or individuals) naturally integrate variable exposures and complex stresses, and are therefore the best overall indicators of aquatic life impact. Biosurveys, for example, provide the most general measure of ecological integrity (water quality and habitat). They can be used to guide planning and management decisions, inventory aquatic resources, describe attainable aquatic life goals, screen and prioritize problem areas, characterize trends, and document the "bottom line" results of control actions. Bioassays, on the other hand, integrate across pollutants and are used more specifically i.e., to discriminate generic toxicity from other types of impacts; and to help interpret narrative "free from" criteria. Finally, tissue residue analyses can be used to identify specific pollutants with concentrations that are either too low or too variable to detect in the ambient medium. All of these tools will be needed to meet the ever increasing demand for meaningful, but economical, monitoring data.

Implementation Issues

Despite the conceptual appeal of broadening the use of bioassessment approaches in water monitoring programs, several practical issues regarding implementation still need to be considered before bioassessments can be effectively implemented on a national scale.

o Biocriteria
  - Do biocriteria necessarily have to be incorporated into water quality standards?

  - Do they have to be quantitative and numerical to be useful?

  - Given an "average" ecoregion, how many and what kinds of evaluations are needed to confirm its boundaries and establish biocriteria? How long does it take and how much does it cost?

  - Are different criteria needed for different types of water bodies; designated uses; different subcommunities; different geographical (e.g., subregional, local) scales; different temporal scales (seasons)?

o Monitoring biocriteria and performing assessments

  - Would methods used to assess criteria differ from those used to develop criteria? If so, why?

  - Should any nonbiological parameters be routinely monitored in conjunction with a bioassessment?

- What is the role of biocriteria in assessing toxics? Habitat degradation?

These are only a few of the issues that will need to be considered before bioassessments can be effectively implemented on a national scale.

Literature Cited

U.S. Environmental Protection Agency. 1987. Surface Water Monitoring: A Framework for Change. U.S. Environmental Protection Agency, Office of Water and Office of Planning and Procedure, Washington, D.C.

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IMPLEMENTATION OF BIOLOGICAL STANDARDS AND CRITERIA IN MAINE'S WATER CLASSIFICATION LAW

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Abstract

Maine has established statutory biological standards in its water classification. This was done with the intent of establishing a set of impact standards which directly measure the biological integrity of the water, a stated goal of both federal and state law. By using the biological standards and associated criteria in a planning role, much of the constraints on use and language, which might be imposed in a regulatory system were avoided. This allowed for definitions and criteria to be written from a technical-scientific perspective, and also allowed greater discretionary use of professional judgement in making biological evaluations. Maine's biological program is created with a set of three narrative standards in its law which range from that sufficient to attain the interim fishable/swimmable goals of the federal act to full maintenance of integrity in a natural status. These narrative standards are further defined in statute with a set of scientific definitions for terms in the standards. These defintions identify specific ecological attributes which may be tested by a hierarchical scheme of tests of descending power and increasing professional judgement to arrive at a decision as to whether a standard is achieved.

Introduction

In the early 1980's, the State of Maine found that its water quality laws were deficient and a new water classification law was passed in 1986 (Maine Revised Statutes Annotated, Title 38, Sections 464 to 468). There were three significant factors which created a need for this change in the law. First, radical improvements had taken place in water quality over much of the state. While improvements had been predicted by water quality models for dissolved oxygen, for instance, it had always been unclear how these improvements would affect the aquatic biota. Within a few years, observation of the reinvasion of many pollution tolerant species was documented. Direct observation could also be made of how differing levels of treatment and loading rates affected the aquatic biota. With this information, the water classification law was found to be deficient in describing the biotic resources of the state.

Secondly, standards and criteria in Maines law did not represent the most current scientific information. In addition to revising standards for dissolved oxygen and enteric bacteria, it was decided that the current knowledge of the aquatic community processes was sufficient to enable application of classification standards and standardized methods and criteria for Maines waters. Use of community assessment is a cost effective measure since it is a direct, holistic evaluation of water quality goals.

Finally, water quality management was evolving in a way which demanded new methods of assessment and more integrated evaluations of water quality. The state is in its third round of licensing. Licenses will

Table 1. Aquatic life classification scheme for Maine's rivers and streams.

Rivers and Streams Classes	Management Perspective	Level of Biological Integrity
AA	High quality water for preservation of recreational and ecological interests. No discharges or impoundments of any kind permitted.	Aquatic life shall be as naturally occurs
A	High quality water with limited human interference. Discharges restricted to noncontact process water or highly treated wastewater of quality equal to or better than the receiving water. Impoundment permitted.	Aquatic life shall be as naturally occurs.
B	Good water quality. Discharges of well treated effluents with ample dilution permitted.	Ambient water quality sufficient to support life stages of all indigenous aquatic species. Only nondetrimental changes in community composition may occur.
C	Lowest quality water. Requirements consistent with interim goals of the Federal Water Quality Act (fishable and swimmable).	Ambient water quality sufficient to support the life stages of all indigenous fish species. Changes in species composition may occur but structure and function of the aquatic community must be maintained.

not be modified unless there is demonstrated impairment of water quality sufficient to affect uses. Former water quality standardswere limited in their ability to detect use impairment. Thus, the biota could offer a feedback mechanism to assess the actual goals for habitat improvement being sought through the licensing system. Water quality management was also evolving through new amendments to the Water Quality Act of 1987 requiring new and added assessment requirements for toxics and nonpoint source pollution as well as traditional assessment requirements. Because toxics and nonpoint source assessments often involve compound pollutants and complex interactions, the biota can lend new insight into the effects of the pollutants.

In order to change Maines water quality program to place significant emphasis on biological assessment, systematic accountability had to be provided. First, a basis in state and federal law for the use of biological information in water quality classification had to be established. The law also needed to be understandable to the public and most notably the legislature. Secondly, demonstration of administrative accountability had to be established. A new biological program had to contribute needed information that other standards and criteria could not provide. The standards also had to be logistically.

Table 2. Definitions of terms appropriate for establishing water quality criteria.

Term	Definition
As naturally occurs	Conditions with essentially the same physical, chemical and biological characteristics as found in situations with similar habitats free of measurable effects of human activity.
Community	Mechanisms of uptake, storage and transfer of function life-sustaining materials available to a biological community which determines the efficiency of use and the amount of export of the materials from the community.
Community structure	The organization of a biological community based on numbers of individuals within different taxonomic groups and the proportion each taxonomic group represents of the total community.
Indigenous	Supported in a reach of water or known to have been supported according to historical records compiled by State and Federal agencies or published scientific literature.
Natural	Living in, or as if in, a state of nature not measurably affected by human activity.
Resident biological community	Aquatic life expected to exist in a habitat which is free from the influence of the discharge of any pollutant. This shall be established by accepted biomonitoring techniques.
Unimpaired	Without a diminished capacity to support aquatic life.
Without detrimental changes in the resident biological community	No significant loss of species or excessive dominance by any species or group of species attributable to human activity.

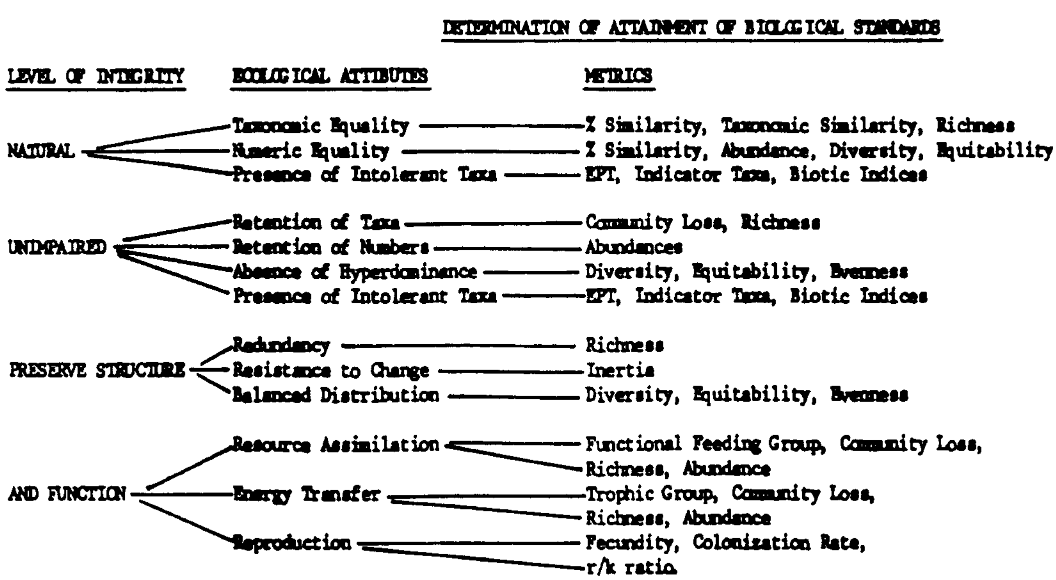

Fig. 1. Determination of Attainment of biological standards.

Finally, there had to be scientific accountability. The biological standards and criteria had to have a solid basis in ecological principales. Field methods and analytical techniques had to be reproducible and accurate.

Development of Biological Standards and Criteria-- Four general questions need to be addressed to provide accountability to a system of biological standards and criteria: 1). what are the goals and purposes, 2). how will the biological standards and criteria be used to achieve those goals and purposes, 3). how can goals be defined biologically, and 4). what sort of decision process is appropriate for biological information.

Goals and Purposes: One of the goals of the Federal Water Quality Act, stated in section 101, is to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters". The problem is to define integrity. All waters, even the most polluted have integrity. Therefore, one must examine the Act further to find what is an allowable range for integrity. Certainly, one standard for integrity is conditions which would be found in waters having no discharges, since another goal of the Act is to eliminate all discharges. A second standard for integrity may be found in the interim goals of the Act which requires water sufficient "for the protection and propagation of fish, shellfish, and wildlife".

Within these bounds set by the Water Quality Act, Maine has established three levels of integrity for flowing freshwaters in its water classification law (Table 1). Class AA and A standards require the biological community to be "as naturally occurs". This is analogous to conditions found without discharges. Class B standards require the aquatic community to be unimpaired by water quality conditions. Discharges are allowed, however, they must only result in changes to the community regarded as benign (e.g. recruitment of new species, increased numbers). All indigenous species must be supported and this typically occurs where nontoxic effluents are discharged into waters with ample dilution. Class C standards require that the structure and function of the aquatic community must be protected. There may be considerable replacement of pollution tolerant species, by tolerant species in Class C waters. All indigenous fish species must be supported by water quality, however, they are not required to be present in a given water body if other factors of habitat or biological interaction preclude their establishment. Class C standards in Maine law are considered analogous to the interim goals of the federal act. Tests for attainment of classification are based on effluent toxicity tests to determine support for indigenous organisms in Class B and C, and measurements of the ambient macroinvertebrate community to determine the status of the resident biological community.

Uses for biological standards: Water quality standards may be used in two ways, a regulatory approach or a planning approach. The regulatory approach is traditional and uses performance standards to regulate selected outputs (e.g. dissolved oxygen). They focus on a single pollutant, are simple to

Table 3. Criteria key to attainment of class A level of integrity "as naturally occurs" (taxonomic equality, numerical equality, presence of intolerant taxa).

1. Percent similarity > [0.7]..A
   [0.7] percent similarity > [0.3]..2
   Percent similarity < [0.3]..............................non attainment.(NA)
   Comparison not possible..6

2. Taxonomic similarity > [0.8]...3
   Taxonomic similarity < [0.8] and [0.6]...................................4
   Taxonomic similarity < [0.6]..NA

3. Percent similarity of * dominant taxa > [0.7].............................A
   Percent similarity of dominant taxa < [0.7] and [0.5]...................4
   Percent similarity of dominant taxa < [0.5]...............................NA

4. Taxonomic similarity of dominant taxa > [0.9]..............................5
   Taxonomic similarity of dominant taxa < [0.9] but may be attributable
   to natural habitat differences **...6
   Taxonomic similarity of dominant taxa < [0.9], but habitat similar.....NA

5. Community richness, diversity, and total abundance are all ± [0.8],
   of reference community...A
   Community richness, diversity, and total abundance ± [0.6 to 0.8]
   of reference community....................................Indeterminant
   Community not as above..NA

6. Ephemeroptera, Plecoptera and Trichoptera all present and EPT
   richness > Diptera richness..7
   Ephemeroptera and Trichoptera present...9
   Not as above..NA

7. Diversity > [3.0]...A
   Diversity < [3.0]...8

8. Equitability > [0.6]..A
   [0.6] > Equitability > [0.3].................................Indeterminant
   Equitability < [0.6]..NA

9. Ephemeroptera and Trichoptera compose at least [50%] of
   dominant taxa...7
   Ephemeroptera and Trichoptera compose less than 50% of dominant taxa...NA

* Dominant taxa are those which compose more than [5%] of total community population.

** Habitat differences exceed ranges recommended in "Methods for Biological Sampling and Analysis of Maine's Waters".

[ ] denotes an undetermined value.

use, good for modeling and enforcement, but are limited in scope and not directly goal oriented. Biological standards are not suitable as performance standards. The planning approach uses impact standards which regulate multivariate outcomes, such as community response. They are an integrative standard which focuses on the state of the resource and are a direct measure of goals. They are not well suited to modeling, are retroactive, and have limited enforcement value. Impact standards provide the manager with a direct means to evaluate the progress of water quality improvements gained through the implementation of various programs (e.g. NPDES, construction grants, nonpoint source).

Definitions of Biological Standards: Integrity may have a multitude of definitions, however, the Federal Water Quality Act may be interpreted as having bounds on the extent of allowable degradation. Within these bounds, Maine has established three narrative biological standards of integrity. These narrative standards must be further refined by establishing appropriate ecological attributes specifically suited to each standard. In Maine, this was done in statute through a set of definitions, which define critical terms in each standard (Table 2). It is important that each definition be ecologically sound. By identifying ecological attributes uniquely associated with each standard, specific metrics can be identified for use in the development of criteria (Fig. 1). For example, the term "as naturally occurs" is defined as conditions with essentially the same physical, chemical, and biological characteristics as found in situations with similar habitats free of measurable effects of human activity. From this definition, it is apparent that various tests of similarity are most appropriate for testing integrity in Class AA and A waters. Criteria are developed for each class based on metrics sensitive to the ecological attributes associated with the standard and will vary across classes.

Decision Process: Maine's water classification statute is explicit in setting biological standards and defining the terms in those standards. From these definitions, an array of metrics can be identified. It is important that these metrics be used in a consistent manner to provide the most reliable assessment. To do this, the metrics are used in a hierarchical sequence using the most powerful metrics first, and relying on secondary tests when the primary tests do not yield clear results. Ecological evaluation is known for large variability in results. To take this into account, Maine's system of criteria evaluation uses a series of trichotomous tests (Table 3). Where results of a metric show a strong pass or fail value, that result is considered valid. Where results are in between, and significance of a particular value is not clear, the hierarchical sequence moves on to other metrics, which test components of the first test, or provide other information about the status of the community. The hierarchical sequence also allows for use of professional judgement and escapes where samples are found to be nonrepresentative due to influences of sampling methods, habitat differences or other factors not associated with water quality.

THE ROLE OF BIOLOGICAL DATA IN WATER QUALITY ASSESSMENT

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Abstract

Although the principal goal of the Water Quality Act is to restore and maintain chemical, physical, and biological integrity, the methods by which regulatory agencies have been attempting to achieve it are primarily chemical and toxicological. Difficulties with defining an ecological approach to assessing biotic integrity have probably led to this reliance on surrogate measures. One purpose of this volume is to define biotic integrity as a practical and workable concept upon which objective biological criteria can be based. Thus compliance with a major directive of the Water Quality Act can be measured directly. This also responds to a mandate of the Water Quality Act of 1987 for the development of biological monitoring and assessment methods as both a supplement and an alternative to the pollutant-by-pollutant criteria approach for toxic chemicals (Section 308).

This biocriteria approach can also be described as a systems approach in which the focus is on the resource (i.e., aquatic life) and its response to different environmental impacts. This approach permits a variety of different resource management options to be examined and used as a strategy to restore or protect the performance of the resource. In contrast, the current chemical specific/toxicity approach can be characterized as a regulatory approach in which the focus is on specific pollutants with specific rules for discharge being specified. This proposal advocates the complimentary use of both approaches, not one to the exclusion of the other.

The use of biological communities, particularly fish and macroinvertebrates, offers a holistic, systems approach to surface water quality assessment and management. Aquatic organisms not only integrate a variety of environmental influences (chemical, physical, and biological), but complete their life cycles in the water body and as such are continuous monitors of environmental quality. Focusing on major organism groups such as fish and macroinvertebrates represents biological evaluation at the sub-community level. This differs from past biological monitoring protocols which advocated the resource intensive monitoring of a variety of different organism groups (e.g., algae, macrophytes, zooplankton, diatoms, etc., in addition to fish or macroinvertebrates) at the same time. Another attractive feature of the biocriteria approach is that sampling need not be conducted under absolute worst case or critical conditions (i.e., $Q_{7,10}$ flow) to determine attainment/non-attainment of aquatic life uses. This certainly presents a powerful assessment tool compared to the steady state

approaches inherent to commonly applied chemical specific and toxicity methods. Including this type of biological field assessment along with traditional chemical and toxicity tools can significantly enhance decision making and regulatory resource allocation, particularly with complex issues.

The type of biological field assessments advocated by this document (i.e., sub-community level analysis) is cost competitive with chemical specific and toxicity testing methods. It is also equally cost effective when the power of the information derived from each is considered. The cost analysis presented in this document tends to refute the widely-held reputation of biological surveys as being prohibitively expensive.

Biological criteria were developed for Ohio rivers and streams using the biosurvey/ecoregion approach and the design of the Stream Regionalization Project in conjunction with the U.S. EPA Environmental Research Laboratory - Corvallis. A set of least impacted reference sites from across the state and within each of the five ecoregions of Ohio were carefully selected and sampled for fish and macroinvertebrates. These sites represent watersheds with the least disturbance from human activity within each ecoregion. Based on these results criteria for three biological indices, the Index of Biotic Integrity (IBI, based on fish), the Modified Index of Well-Being (I_{wb}, fish), and the Invertebrate Community Index (ICI, macroinvertebrates) were derived. This design satisfies the definition of biological integrity as the biological performance achieved by the natural habitats within a region.

Practical uses of this approach include determining appropriate and attainable aquatic life uses for surface waters, extending antidegradation concerns to nonpoint and habitat impacts, enhanced problem discovery for toxics, prioritizing the use of regulatory resources (e.g., permits, grants, 304(l) lists), and as a check on the attainment of Water Quality Act goals (e.g., 305(b) reporting).

Several examples from past Ohio EPA biological surveys are presented as a demonstration of how the biological criteria can be used and the complex combination of point source, nonpoint source, and habitat factors that are common to most study areas. The problem discovery capabilities of biological assessment are emphasized.

Previously Published as:

Ohio Environmental Protection Agency. 1987. Biological criteria for the protection of aquatic life: Volume I. The role of biological data in water quality assessment. Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio.

THE DEVELOPMENT OF FISH POPULATION-BASED BIOCRITERIA IN VERMONT

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Abstract

The Vermont Department of Environmental Conservation is presently modifying two fish population-level indicies for potential use as biocriteria in permit compliance and stream classification. Modifications of Karr's Index of Biotic Integrity and Pinkham and Pearson's similarity coefficient (PPSC) were selected for use in defining the water quality standard, "undue adverse effect on the aquatic biota". A modification of Karr's IBI for the Northeast by Miller et al. provided a starting point in adopting the IBI for Vermont's species depauperate wadeable streams. Omernick's ecoregion format was used to establish ecoregional species richness standards used by the Vermont version. The PPCS was modified to more heavily weigh contrasts between the more dominant species. The calibration of both indicies is in progress. The IBI has been applied to data from 44 sites on 28 streams while the PPCS has been applied at eight sites. The Vermont IBI has not responded fully to every type of community disturbance, i.e. flow regulation and some toxins. Aside, however, from being a highly integrative index, scoring of the IBI does provide a framework of metric assessment permitting analysis of individual community attributes; an advantage in applying professional biological judgement. The PPCS appears to be sensitive to any shift in compositional and abundance changes, but computation of the index value reveals little descriptive information on either contrasted community. The potential weakness of each index seems to be compensated for by the other when used concurrently in control-test comparisons.

Introduction

Specific biocriteria have been proposed by the Vermont Department of Environmental Conservation which use macroinvertebrate communities in determining in-stream compliance of indirect dischargers through the Indirect Discharge Program. A macroinvertebrate sampling and analysis protocol is in place which defines "significant impact to the aquatic biota" in making this determination. Since activities leading to the development of appropriate fish population descriptors have taken place at a slower rate, formally proposed fish-based biocriteria have yet to be presented. The overall objective of this effort is to generate a systematic method of evaluating the integrity of the fish community which can be utilized in an analogous manner to the macroinvertebrates protocol but which defines the less vigorous Water Quality narrative criterion, "undue adverse effect to the aquatic biota". The Department also recognizes the potential for fish population assessment in monitoring and stream classification programs. This manuscript describes the process by which two biological indicies were selected and are being modified for use on fish communities in wadeable streams in Vermont. This effort can be partitioned into two steps: 1) selection and verification of indicies, and 2) integration of these indicies into compliance and monitoring programs as biocriteria.

Table 1. Considerations in Developing Fish Population Biocriteria in Vermont.

Specific Index Requirements

1. Must measure integrity of entire fish population;
2. Must accurately measure fish population responses to physical habitat degradation as well as point and nonpoint water quality impacts;
3. Must have low variability of known quantity;
4. Must be applicable to a variety of stream habitats.

Data Quality Requirements

1. Must develop standardized physical habitat assessment method for site comparison;
2. Must establish sampling protocols
   a. Sampling method
   b. Sampling effort
   c. Site selection

Biocriteria Considerations

1. Relate index results to narrative criteria (integrate use of index into Water Quality Standards).
2. Must systematically establish professional judgement as an input to decision making process.

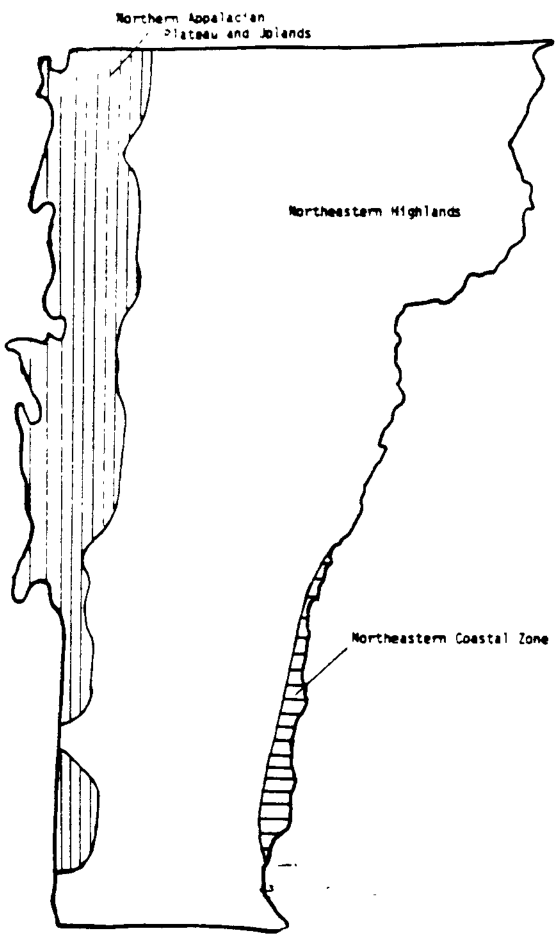
Modifications of Karr's (1981) Index of Biotic Integrity (IBI) and the similarity coefficient of Pinkham and Pearson (PPCS) (1976) were selected for use. The verification and calibration effort for both indicies is still in progress. To date, the IBI has been applied to 76 sites on 43 streams. No in-depth evaluation of our results will be presented here due to the incomplete data base. The discussion will include the rationale used to modify both indicies, some general interim results and finally, information needs to be addressed.

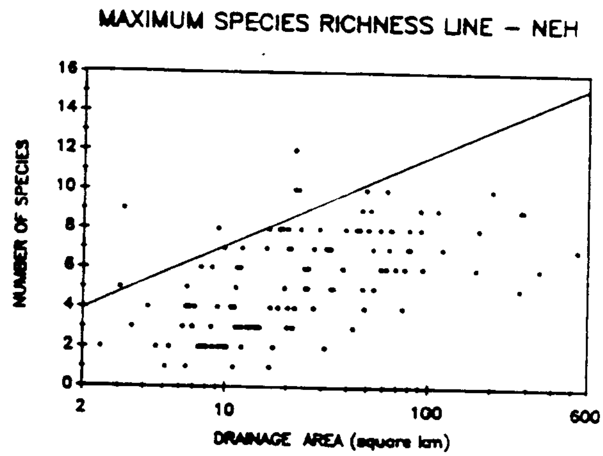
Methods and Materials

The development of fish population-based biocriteria began in 1986. In the first two years data used in the index testing often originated from other sampling programs. For 1988, however, significantly more time has been allotted specifically for IBI-PPCS verification. While general goals were defined at the onset of these activities, specific objectives and concerns for data requirements evolved as the work progressed. Future efforts will address specific information needs that were generated from past work. The principle data requirements and general concerns appear in Table 1.

Vermont Stream Populations

In characterizing fish community attributes of wadeable streams, historical data was organized by ecoregion (Omernick 1987). Vermont, a small state, contains only three ecoregions, one of which, the Northeastern Coastal Zone appears to include too small an area to be treated separately (Figure 1). Most of the state is covered by the Northeastern Highlands (NEH). This ecoregion, characterized by relatively high elevations, includes the Green Mountains, the Vermont Piedmont and the (Vermont)





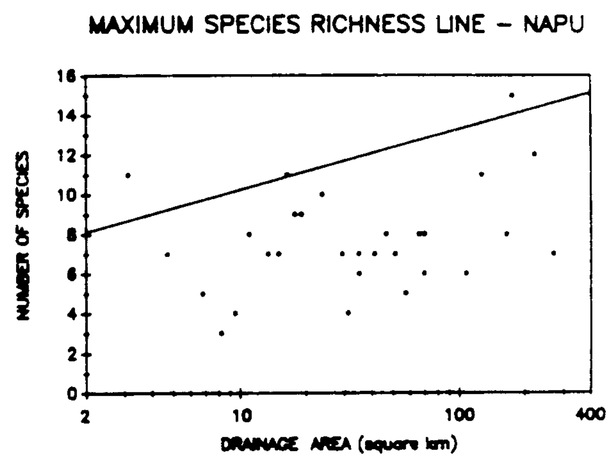


Fig. 1. Ecoregions of Vermont from Omernik 1987.

Fig. 2. Maximum species richness lines for NEH.

Fig. 3. Maximum species richness lines for NAPU.

Northeastern highlands. The other major ecoregion, the Northern Appalachian Plateau and Uplands (NAPU) span the Eastern third of the state, including the Champlain Valley and lower elevations of the Taconic Mountains.

Available data indicate that Vermont stream communities are relatively species-depauperate with most streams supporting fewer than ten species. Figures 2 and 3 plot species richness by sampling site drainage area for both major ecoregions. Streams in the NEH are generally dominated by insectivores. Headwater reaches often contain only brook trout. Species additions, progressing downstream, commonly include slimy sculpin, blacknose and longnose dace followed by creek chub, white suckers, fallfish, brown and rainbow trout. A small number of additional cyprinids, tesselated darter and one to three centrarchids may complete the community in the lower reaches of

Table 2. A preliminary IBI for Vermont.

	Scoring Criteria 5	3	1
Species Richness and Composition			
1. Total number of fish species	Maximum species richness lines		
2. Number and Identity of Tolerant Species	>1	1	0
3. Number and Identity of Benthic Insectivores	>2	1-2	0
4. Proportion of Individuals as White Sucker	<10%	10-25%	>25%
Trophic Composition			
5. Proportion of Individuals as Generalist Feeders	<20%	20-45%	>45%
6. Proportion of Individuals as Insectivores	>65%	30-65%	<30%
7. Proportion of Individuals as Top Carnivores: Cold water	>10%	3-10%	<3%
Warm water	>5%	1-5%	<1%
Fish Abundance and Condition			
8. Abundance in Sample	moderate to high	low	very low
9. Proportion of individuals with disease, tumors, fin damage and other anomalies	0-1%	1-3%	>3%

Excellent	43-45
Good	36-39
Fair	29-33
Poor	22-25
Very Poor	9-19

larger streams. Species richness in NAPU streams appears to be slightly greater than in NEH streams of similar drainage area. The total species list from NAPU streams includes most species from the NEH plus an additional fifteen species (mostly cyprinids and darters) not found int he NEH streams. Data collected to date suggests that longitudinal species addition seems to occur at a higher rate in NAPU streams. Most streams of this ecoregion support warmwater populations, devoid of trout.

The Vermont IBI

It was recognized early that if the IBI concept was to be applied to Vermont's streams that extensive modification of the midwest original (Karr 1981) would be required. Following a review of two IBI modifications from Karr's original for Eastern streams, the modification by Miller et al. (unpublished manuscript) for Merrimack (New Hampshire) and Connecticut (Massachusetts) drainages was selected as a starting point. The present nine-metric Vermont IBI (Table 2) contains eight of twelve metrics from Miller et al. Some of these metrics were rescored or modified. One metric was taken from the modification of Leonard and Orth (1986).

An IBI is applied by assigning a score of 5, 3 or 1 to each metric. A score of "5" denotes full agreement with conditions from a relatively unimpacted site while a "1" represents the greatest deviation from that expected. A score of "3" reflects an intermediate level of deviation. Metric scores are summed, with the resultant value placed into a qualitative category ranging from very poor (low score) to excellent

Table 3. Fish Species occurring in Vermont Streams considered as intolerant to general habitat and Water Quality degradation based on published literature accounts.

Brook trout	Chain pickeral
Brown trout	Cutlips minnow
Slimy sculpin	Northern Redbelly dace
Blackchin shiner	Silvery minnow
Blacknose shiner	

(high score). A brief discussion of the rationale for each metric from the Vermont IBI follows.

Metric 1. Total Number of Species. Lines of maximum species richness were generated from historical data from 154 streams sampled by the Vermont Departments of Fish and Wildlife and Environmental Conservation (Figures 1 and 2). These lines represent ecoregional standards. Following Karr's (1981) methods, a fit-to-eye line was drawn to include 95% of the data and to follow the general slope of the plot. Two other lines, approximately trisecting 95% of the data below the maximum species richness were scored according to Karr. A general, though not well developed, trend of increasing number of species with stream drainage area was observed for both ecoregions. This metric appears only to be sensitive at moderate to severe levels of degradation as reflected by the present Vermont IBI data base.

Metric 2. Number and Identity of Intolerant Species. This metric is often scored by the use of a line of maximum species richness (Karr 1981; Miller et al. unpublished). Since species richness in Vermont streams is low, any variation in the numbers of intolerant species expected between sites and ecoregions has not been detected as yet. As a result, one set of scores has been assigned to all streams. Eleven species have been classified as intolerant based on the available literature (Table 3).

Metric 3. Number and Identity of Benthic Insectivores. Since stream habitat degradation may represent the greatest threat to aquatic biota in Vermont, the inclusion of metrics sensitive to a wide breadth of feeding preferences is of particular importance. An unstable benthic macroinvertebrate community in the presence of degraded conditions threatens those fish species which rely on that community as a primary food base (Karr 1986). Insectivores dominate fish communities in healthy streams in both Vermont ecoregions. As with metric 2, no variation between sites or ecoregions has been observed and one set of scores has been assigned to all streams. A typical undisturbed stream supports from one to three benthic insectivores. Trophic classification follows the available literature (Table 4).

Metric 4. Proportion of Individuals as White Sucker. This species was selected due to its ubiqutous distribution in both ecoregions. The white sucker is commonly regarded as tolerant to many forms of degradation (Trautman 1981; Twomey 1984). As generalists feeders (Miller et al. unpublished; Leonard and Orth 1986) they are better suited to a shifting food base in the presence of degraded conditions than are more specialized feeders (Karr et al. 1986). Thus far white sucker have only occurred in higher densities in degraded sites. This metric follows the substitution by Miller et al. of white sucker for Karr's green sunfish metric as a tolerant species.

Metric 5. Proportion of Individuals as Genralist Feeders. Leonard and Orth (1986) substituted this metric for Karr's omnivore metric because 1). the omnivore classification was believed too restrictive in defining species which were able to shift food habits in response to a variable food base, and 2). some generalist feeders, i.e. creek chub were not classified as omnivores yet, were very tolerant to many forms of perturbation. Use of the omnivore classification resulted in a conflict in scoring metrics (and a less responsive index). The placement of creek chub and fallfish into the generalized feeder category with true omnivores appears to be appropriate in that *Semotilus* in Vermont streams is generally observed as a dominant only in degraded stream reaches. This metric will usually vary inversely in scoring with metrics 3, 6 and 7.

Metric 6. Proportion of Individuals as Insectivores. Miller et al. substituted this metric for Karr's insectivorous cyprinids metric due to the paucity of insectivorous cyprinid species in streams of the Northeast. This was also deemed a reasonable substitution for Vermont streams. This metric is comparable in function to metric 3 (benthic insectivores species) but includes surface and midwater feeders as well.

Metric 7. Proportion of Individuals as Top Carnivores. This metric is analogous to the top level carnivore metric of Miller et al. and others. Since a significant portion of streams in Vermont support naturally reproducing trout, the three trout species (as well as burbot) are included as top carnivores. Since unimpacted wadeable streams appear to contain trout and warmwater piscivores at different densities, two scoring ranges have been established. For sites represented by both groups, the group scoring the highest will be represented in the metric. The modification of Miller et al. excluded from consideration upland coldwater sites which support trout. The author does not believe that the presence of trout and a low number of other species at a site precludes application of an IBI. It is believed that enough information exists to accurately score the IBI if a generalist feeder and at least three other non-salmonid species are present. This condition represents a proposed minimum criterion for applying the Vermont IBI.

Metric 8. Abundance of Sample. More data is presently needed to calibrate this metric. Since a wide range of productivity exist in Vermont streams and since yearly variation in this parameter is high, this metric will probably be scored conservatively. Thus far, as Karr et al. (1986) recommends, catch per unit effort (CPUE) has been used in scoring this metric.

Table 4. Trophic Classification of Vermont's Stream Fishes. Determinations are based on the published literature.

TOP CARNIVORE

Chain Pickeral (Esox niger)
Northern Pike (Esox lucius)
Largemouth Bass (Micropterus salmoides)
Smallmouth Bass (Micropterus dolomieui)
Rock Bass (Ambloplites rupestris)
Brook Trout (Salvelinus fontinalis)
Brown Trout (Salmo trutta)
Rainbow Trout (Salmo gairdneri)
Burbot (Lota lota)

BENTHIC INSECTIVORES

Blacknose Dace (Rhinichthys atratulus)
Longnose Dace (Rhinichthys cataractae)
Cutlips Minnow (Exoglossum maxillingua)
Slimy Sculpin (Cottus cognatus)
Mottled Sculpin (Cottus bairdi)
Shorthead Redhorse (Moxostoma macrolepidotum)
Eastern Sand Darter (Ammocrypta pellucida)
Tessellated Darter (Etheostoma olmstedi)
Logperch (Percina caprodes)

INSECTIVORE

Blackchin Shiner (Notropis heterodon)
Emerald Shiner (Notropis atherinoides)
Rosyface Shiner (Notropis rubellus)
Spotfin Shiner (Notropis spilopterus)
Spottail Shiner (Notropis hudsonius)
Blacknose Dace (Rhinichthys atratulus)
Longnose Dace (Rhinichthys cataractae)
Cutlips Minnow (Exoglossum maxillingua)
Finescale Dace (Phoxinus neogaeus)
Bluegill (Lepomis macrochirus)
Pumpkinseed (Lepomis gibbosus)
Redbreast Sunfish (Lepomis auritus)
Slimy Sculpin (Cottus cognatus)
Mottled Sculpin (Cottus bairdi)
Eastern Sand Darter (Ammocrypta pellucida)
Iowa Darter (Etheostoma exile)
Tessellated Darter (Etheostoma olmstedi)
Logperch (Percina caprodes)
Yellow Perch (Perca flavescens)
Shorthead Redhorse (Moxostoma macrolepidotum)
Banded Killifish (Fundulus diaphanus)
Brook Stickleback (Culaea inconstans)
Trout-perch (Percopsis omiscomaycus)

GENERALIZED FEEDER

Blacknose Shiner (Notropis heterolepis)
Bluntnose Minnow (Pimephales notatus)
Common Carp (Cyprinus carpio)
Common Shiner (Notropis cornutus)
Creek Chub (Semotilus atromaculatus)
Fallfish (Semotilus corporalis)
Fathead Minnow (Pimephales promelas)
Golden Shiner (Notemigonus crysoleucas)
Lake Club (Couesius plumbeus)
Mimic Shiner (Notropis volucellus)
Northern Redbelly Dace (Phoxinus eos)
Pearl Dace (Semotilus margarita)
Sand Shiner (Notropis stramineus)
Eastern Silver Minnow (Hybognathus regius)
Black Bullhead (Ictalurus melas)
Brown Bullhead (Ictalurus nebulosus)
Stonecat (Noturus flavus)
Longnose Sucker (Catastomus catastomus)
White Sucker (Catastomus commersoni)
Fantail Darter (Etheostoma flabellare)
Mudminnow (Umbra limi)

Metric 9. Proportion of Individuals with Disease, Tumors, Damage and Other Anomalies. This metric has a relatively narrow range of application in Vermont as it is sensitive to only severe degradation (Karr et al. 1986). The most common anomaly thus far is heavily infestations of black spot (Neascus sp.). Steedman (1988) substituted the occurrance of black spot alone for Karr's original metric, as this was the predominant anomaly in streams in the Toronto area.

Three metrics from the modification of Miller et al. were not used in the Vermont IBI.

Metric 2. Number and Identification of Native Water Column Species. This is a substitute metric for Karr's original number of sunfish species metric. It was not included in the Vermont IBI because of the probable conflict in scoring with the generalist feeder metric. many water column species, i.e. creek chub, fallfish, common shiner and golden shiner) are omnivores and generalist insectivores. The two metrics then would most likely cancel each other by scoring, in opposite directions, a species which is both opportunistic and a water column feeder. Low species richness in Vermont streams may also be responsible for the preclusion of the water column feeder metric.

Metric 4. Number and Identity of Sucker Species. Only two sucker species are known to inhabit wadeable streams in Vermont. While the white sucker is generally regarded as tolerant to many forms of degradation, the longnose sucker is believed to have a narrower range of habitat tolerances

Table 5. An example of the PPCS (Pinkham and Pearson 1976) (A) and the weighted modification of that version (B).

A.

	Abundance		
	Site A	Site B	Quotients
Species A	100	75	75/100 = 0.75
B	10	50	10/50 = 0.20
C	1	10	1/10 = 0.10
D	1	0	1/0 = 0
			1.05/4 = 0.26 = B

B.

	Abundance				
	Site A	Site B	Quotients	Factor	Quotients
Species A	100	75	75/100 = 0.75 x	2.00 =	1.50
B	10	50	10/50 = 0.20 x	1.25 =	0.25
C	1	10	1/10 = 0.10 x	1.00 =	0.10
D	1	0	(not included)	4.25	1.85
			4.25/1.85 = 0.44 = B		

(Edwards 1983). Karr's intent was to equate greater numbers of sucker species (most of which were intolerant) with higher site integrity. Clearly then, this metric would be inappropriate for use in Vermont streams.

Metric 11. Proportion of Individuals as Hybrids. To date, few hybrids have been identified in Vermont streams. A problem exists in the accurate field identification of hybrid cyprinids, the group in Vermont most likely to exhibit this phenomenon.

A condition of the modification of Miller et al. excluded exotic species from the scoring of all but one metric. Exotics were viewed as part of the degradation. Because of a general lack of severly impacted sites combined with the existence of physical barriers prohibiting extensive upstream movement, exotic non-salmonid species do not comprise a significant component of wadeable streams in Vermont. Exotic trout species (brown and rainbow) are included in the scoring of the Vermont IBI under the following conditions: 1) the site or reach sampled can support natural reproduction of those species, and 2) sampling to take place at a location and time, that is enough removed from (1 km, 3-4 months) from the stocking site and time.

Some Proposed Guidelines for the Application of the Vermont IBI

A minimum criterion of four non-salmonid species, including a generalist feeder, must be met to apply this index. In control-test site comparisons this prerequisite applies to the control populations only. All sampling must be conducted between mid-August and mid-October. This period corresponds to the yearly low flow period permitting most efficient sampling. Since small fish are not as vulnerable to electrofishing gear (Nielson and Johnson 1985) only fish larger than 25 mm total length will be included in the index. Stocked Atlantic salmon fingerlings will be excluded from index consideration due to their inability (as yet) to spawn in streams. Abundance of fingerlings on certain reaches can be high due to high stocking densities and absence of sportfishing mortality.

Fishing effort required remains a less-easily defined guideline. Minimum effort in the sampled reach has not yet been determined. The choice exists as one between a one to two sweep CPUE and the multiple-sweep population estimator of Carle and Strub (1978).

Physical habitat conditions have been shown to be an important determinant of fish distribution (Gorman and Karr 1979; Horowitz 1978; Schlosser 1982). Hendrick et al. (1980) identified the process of selecting control and test sites that are similar in habitat as "one of the most difficult problems encountered in the biomonitoring of fish". While trained professionals can often select two sites, like in habitat (Hendricks et al. 1980), it is believed that a form of documentation of that likeness is necessary. A systematic method of habitat analysis then, will likely be required for all biocriteria related population sampling. This analysis will provide a measure of habitat similarity between control and test sites. This measure is of importance in cases where: 1) water quality impairment is suspected and differences in physical habitat are to be minimized, and 2) changes in habitat are to be documented at sites where a physical habitat-related impairment is suspected.

Although the quantity of area sampled varies in published studies, most investigators strive to sample all major habitat types in attempting to produce a fully representative sample (i.e. Mahon 1980; Berkman et al. 1986; Larson et al. 1980; Leonard and Orth 1986; Steedman 1988). Karr et al. (1986) suggests a minimum length of 100 m for structurally simple streams. For larger, more complex (habitat diverse) streams, Karr et al. (1986) suggests a minimum of two habitat cycles. Leonard and Orth (1986) included two habitat cycles in their 50 m length sites in small West Virginia streams. Hankin (1986) stressed the importance of sampling with regard to habitat type rather than pre-established site length. He maintained that sampling habitat-defined sections minimized errors in estimating abundance when compared to length-defined site estimates. It is likely that a combination of minimum distance and number of habitat cycles will be incorporated into Vermont's sampling requirements.

<u>Index of Biotic Similarity</u>

A modification of Pinkham and Pearson's (1976) PPCS (B) is proposed for concurrent use with the Vermont IBI. Use of the Vermont IBI to date, indicates that it may not be completely responsive to all pertebations, i.e. flow regulation and some toxins. The PPCS appears to be sensitive to any change in species abundance or composition within a community. A disadvantage of the PPCS is that the index value as well as the computation of that value provide little information on the nature of the community itself. Simultaneous use of both indicies would appear to combine the positive aspects of sensitivity (PPCS) and description of community parameters (IBI) into the final biocriterion.

The PPCS produces a measure of similarity (0-total dissimilarity to 1.0-total similarity). For use in water quality standards compliance, the implicit assumption is made that an altered or stressed population will, when contrasted to a control population, exhibit progressively lower values (towards dissimilarity) with increasing population impact.

The PPCS is:

$$\frac{1}{k}\sum_{i=1}\frac{\min Xia, Xib}{\max Xia, Xib}$$

where k = number of comparisons between sites;
x = number of individuals in taxon i;
a,b = Site A, Site B

An example illustrates the computation of the formula in Fig. 5-A.

In its present form, the index weighs contrasts between all species equally, regardless of their dominance in the community. Pinkham and Pearson (1976) stated that in cases where organisms from the same trophic level are to be contrasted, it may be more desireable to weigh each contrast according to the relative abundance of that taxon. This author agrees with that assertion that increased significance should be attributed to changes in the more abundant species. The original un-weighted PPCS has also been shown to be more susceptible to sampling error (Brock 1977). Pinkham and Pearson noted this tendency as well, using the example of: a change of one individual will more profoundly alter the index from a 3-3 to a 3-2 contrast than it would if the abundance were higher: 324-325 to a 323-325. Though Pinkham and Pearson proposed a weighted modifcataion (B_2) it was not selected for use because of its inability to weigh contrasts when one taxon was absent from the pair. The following modifications to the IBS are being proposed by the author:

1. Species used in the paired contrasts should comprise at least 1% of the total population or have a density of at least 50 individuals/ ha (from a catch per unit effort estimate based on two electrofishing sweeps) from at least one site.

2. Species used in the paired contrasts will be weighed according to their abundance at both sites combined, using the following factors:

For species comprising 1-5% of the total, multiply the quotient by 1.0; for 5.1-10%, 1.25; for 10.1-15%, 1.50, for 15.1-20%, 1.75; for 20%, by 2.0 (See Table 5-B for example of application).

From the example demonstrated in Table 5-B, species D was eliminated from the contrast because it did not meet the 1% criterion. The remainder of the species were weighted accordingly, resulting in an increase in the PPCS from 0.26 to 0.44. The value from the modified PPCS would intuitively appear to better represent the true changes between these two hypothetical populations.

Results and Discussion

The process of developing biocriteria in Vermont is in progress. Subsequently, the available data set is too small to present a conclusive discussion of the results. A few general trends have been observed and will be discussed together with specific objectives for on-going work.

The Department has, thus far, focused on extensively testing the Vermont IBI over a number of streams rather than intensively on a few streams. The distribution of IBI scores from 44 sites sampled prior to 1988 is skewed towards the higher values despite an

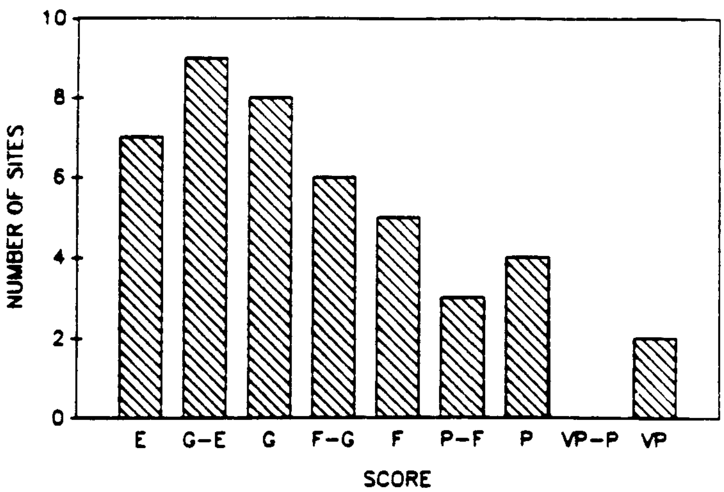


Fig. 4. Distribution of Vermont Index of Biotic Integrity scores for 44 sites.

attempt to include more degraded sites in the testing (Figure 4). Five of the six sites which were rated very poor or poor are known by the Department to be "trouble spots". Substantial water chemistry data exists from four sites which verify the low IBI scores. The Vermont IBI was judged to have fully responded to disturbance from chlorinated wastewater effluents, physical habitat degradation and ammonia toxicity. Sites which scored poor to fair, fair, and fair to good, seem to be exhibiting less definable intermediate impacts from cumulative nonpoint and point sources as well as physical habitat degradation. The six sites rating excellent were all cold water trout streams, five of which are located in the NEH ecoregion. At this point in the testing it appears that streams in the NEH score slightly higher than streams located in the NAPU. It is presently too early to speculate whether this tendency is due to general habitat quality or merely a result of differential index scoring for streams with inherently different trophic composition.

The Vermont IBI has not shown a sensitivity to all types and levels of impacts. Abundance was dramatically reduced (90%) at two sites where the index failed to respond fully. One test site showed excessive BOD and chlorine levels while the other contained high levels of copper from mine drainage. A third site was exposed to routine dewatering from an upstream hydrogeneration facility. Below the facility all major species were present, however, overall abundance was reduced nearly 50%. The Vermont IBI was only 6 points lower at the impacted site indicating "good" conditions. The omission of three of Miller et al.'s "original" metrics is not considered responsible for these inconsistencies in the Vermont IBI. To determine this, the modification of Miller et al. was applied, as well as, the Vermont version plus the three omitted metrics. Neither IBI responded to a greater degree than did the nine-metric Vermont IBI at any of the three cases.

The Department believes the IBI concept to be sound and with potential for use as Biocriteria. The IBI not only integrates several community attributes into a single value, increasing the validity of that value, but through scoring each metric individually, the computation of the IBI also provides the biologist with an opportunity to examine various community attributes separately. This process facilitates the use of professional judgement by the biologist which is considered by the Department to be a vital component of the total site evaluation.

The weighted PPCS has not been tested as extensively as the Vermont IBI. This index has been applied to eight contrasts (two sites each) on eight streams. Values at six degraded sites ranged from 0.02-0.33 while values at two unimpacted replicate sites were 0.56-0.71. For the weighted PPCS the question seems not whether it responds to changes in community integrity, but rather to what degree it responds and how will that translate into final biocriteria?

Information Needs

Further sampling will include a more intensive sample design which will focus on the spatial and temporal factors which effect the scoring of the two indicies. Specific objectives for further sampling program (when data are combined with the past years information) are:

1. To characterize expected variation in both indicies. This will be attempted by sampling a number of replicate sites which will be similar in physical and chemical characteristics.

2. To examine the effects of temporal variability within the low flow period of August to early October. A number of sites will be sampled once between mid-August and early September and again between late September and early October.

3. To better define the sensitivity of both indicies. More sites which vary in extent of degradation will be sampled.

4. To evaluate the effectiveness of using catch per unit effort data (CPUE) in describing abundance.

5. To contrast results from fish population indicies with those from macroinvertebrate populations. Concurrent sampling of fish and macroinvertebrate populations will provide information on how evaluations from each trophic level may be used singly or together in making site evaluations.

An additional concern yet to be addressed specifically, is how to systematically involve professional judgement into specific biocriteria. Critical to this problem is quantifying the role of professional judgement in the decision making process. Prior to the anticipated 1989 completion of the proposed Vermont fish population biocriteria, other issues such as data quality, minimum sampling effort and habitat analysis methodologies will also be addressed.

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THE USE OF FISH COMMUNITIES IN ECOREGION REFERENCE STREAMS TO CHARACTERIZE THE STREAM BIOTA IN ARKANSAS WATERS

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Abstract

The State of Arkansas has been subdivided into six ecoregions based on the homogeneity of land surface forms, potential natural vegetation, soil types and land uses. Within each ecoregion, reference streams of various sizes, excluding the large rivers, and with the least amount of point source and non-point source disturbances were selected for intensive physical, chemical and biological sampling. This data was used to characterize the biotic communities of these streams and establish water quality criteria which will protect all stream uses. Fish communities of the reference streams were distinctly different among the ecoregions. The average number of species collected per sample site was similar among the ecoregions; however, the total number of species collected per ecoregion is notably different. The Arkansas River Valley and the South Central Plains ecoregions have the greatest species richness and the Delta ecoregion is the lowest in species richness. Species of fish sensitive to environmental change comprised 50% or more of the community in the Boston Mountains, Ozark Highlands and Ouachita Mountains ecoregions. Delta ecoregion fish populations contained less than 1% sensitive species. Comparisons of the ten most abundant species from each ecoregion by use of a similarity index shows very little similarity among the ecoregions.

Introduction

The delineation of regions that are distinctly homogeneous has been completed by resource managers for decades in an effort to more efficiently manage a variety of natural resources. Many of the early attempts established physiographic regions based on geographic characteristics, regions of similar vegetation type and regions of various land use patterns. These were all single character classifications with specific needs in mind. Later, in an attempt to characterize ecological relationships, several workers incorporated various combinations of multiple characteristics such as soils, climate, water resources, vegetation, land uses and others into ecoregion classifications (USDA Soil Conservation Service 1981; Bailey 1976; Warren 1979).

Most recently, Hughes and Omernik (1981) and Omernik et al. (1982) proposed methods for development and uses for ecoregions. The potential uses of these ecoregions include: 1) comparisons of land/water relationships within a region; 2) establish realistic water quality standards for regional rather than a large scale application; 3) location of monitoring and reference sites; 4) extrapolate from site specific studies; and 5) predict effects and monitor environmental changes resulting from pollution control activities (Omernik and Gallant 1986).

The ecoregions of Omernik (1987) were developed from four small-scale maps of interrelated land characteristics. These include: land uses, land surface forms, potential natural vegetation and soil types. The regions are

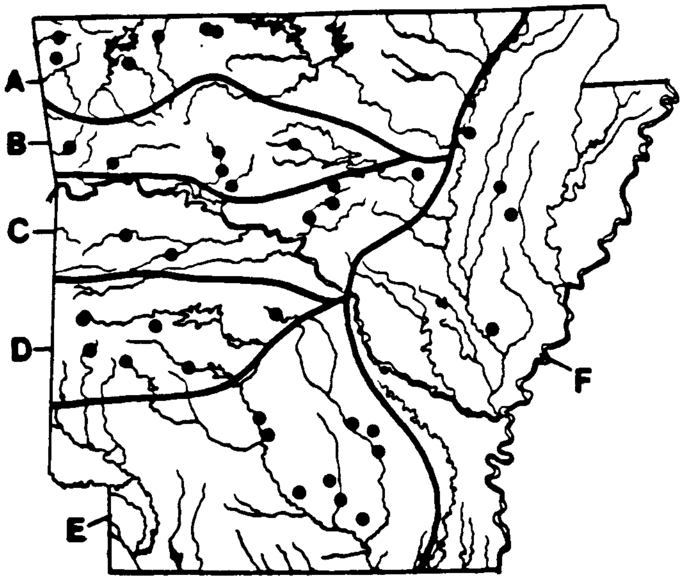


Fig. 1. Reference stream sample sites within Arkansas ecoregions. A-Ozark Highlands, B-Boston Mountains, C-Arkansas River Valley, D-Ouachita Mountains, E-South Central Plains, F-Mississippi Alluvial Plains (Delta).

delineated as the areas of greatest homogeneity. Within each region, the areas which share all of the characteristics that typify the ecoregion are distinguished as the most typical area. Areas which share most but not all of the similar characteristics are designed as generally typical of the region.

The ecoregions within Arkansas and surrounding areas were developed for the U.S. Environmental Protection Agency, Region VI, Dallas and for the Arkansas Department of Pollution Control and Ecology to assist with Arkansas' stream reclassification project. The ecoregions in Arkansas include six distinct regions (Fig. 1): 1) Ozark Highlands; 2) Boston Mountains; 3) Arkansas River Valley; 4) Ouachita Mountains; 5) South Central Plains; 6) Mississippi Alluvial Plain (Delta). These regions are very similar to the natural divisions and sub-divisions of Arkansas as described in Arkansas Natural Area Plan (Foti 1974) and further refined by Pell (1983). The natural divisions of Foti were developed from factors such as: primary vegetation, topography, surface geology, soils and surface hydrology.

Ground reconnaissance and field investigations have resulted in a slight modification of the western segment of the ecoregion boundary between the Arkansas River valley and the Ouachita Mountains from that purposed by Omernik (1987).

Materials and Methods

In order to characterize the physical, chemical and biological features of the biotic environment within each of Arkansas' ecoregions, the Arkansas Department of Pollution Control and Ecology selected a series of streams of varying sizes within each ecoregion for detailed investigation. These reference streams were selected, where possible, within the most typical area of the ecoregion, and only streams with the least amount of point and non-point source disturbances were chosen. A sample site on each stream was established, and both low-flow, high-temperature summertime and steady-state flow, springtime sampling was done. The sampling included detailed measurements of the physical features of the stream, analysis of 18 water quality parameters, a 72 hr continuous record of dissolved oxygen and water temperature, intensive sampling of the stream macroinvertebrate population and a comprehensive fish population sample.

The summer fish population sampling was done with the fish toxicant rotenone or with electrofishing devices. Most of the spring sampling was done with trammel nets of mesh sizes from 2.5 to 8.9 cm. Spring fish sampling was to identify migratory fishes in the area and verify fish spawning activities. The summer sampling identified the total resident fish population and established the relative abundance of each species.

Sample sites with very small or no flow, with reduced visibility into the water and with numerous instream obstructions were sampled with rotenone. If flow existed at these sites, a block net was utilized at the downstream limit of the sample area and rotenone was detoxified with potassium permanganate below the sample area. Areas sampled ranged from about 0.1 to 0.4 ha.

Electrofishing gear was used at sites which had substantial flow, high visibility into the water and where much of the stream could be waded by workers in chest-waders. A gasoline powered generator with 3500 watt AC output was used as a power source. The electrodes were energized directly from the generator. Swift flowing riffle areas were blocked with a seine and stunned fish were allowed to drift into the seine. Sampling was conducted in an upstream direction and the sample areas were usually from 0.4 to 1.6 km in length. All areas that could be efficiently worked were sampled until it became apparent that all existing habitats had been sampled and the fish species and their relative abundance was well established by the sample.

All possible fishes were dipped from the water and preserved in 10% formalin for later identification and enumeration. When large numbers of the same species were observed while electrofishing, only an occasional "dip" sub-sample was made but notes on the species abundance were recorded. Each fish species from all summer samples was given a relative abundance value as described in Table 1.

These values were determined from the number of fish in each species size group, field observations of fishes which were not collected, general knowledge of fish species life-history, selectivity of the sample gear and limitations existing at the sample site. No extensive efforts were made to determine an accurate separation of the young and intermediate age groups of each species. Such determinations were based on the presence or absence of a variety of distinctive size groups. All calculations of total community percentage were made with the relative abundance values.

A list of sensitive fishes for Arkansas were developed from a consensus of six ichthyologists who were familiar with Arkansas fishes and their habitat. Each was asked to designate the species they believed to be intolerant of moderate environmental changes

Table 1. Criteria for assigning relative abundance values to species and age group of fishes collected.

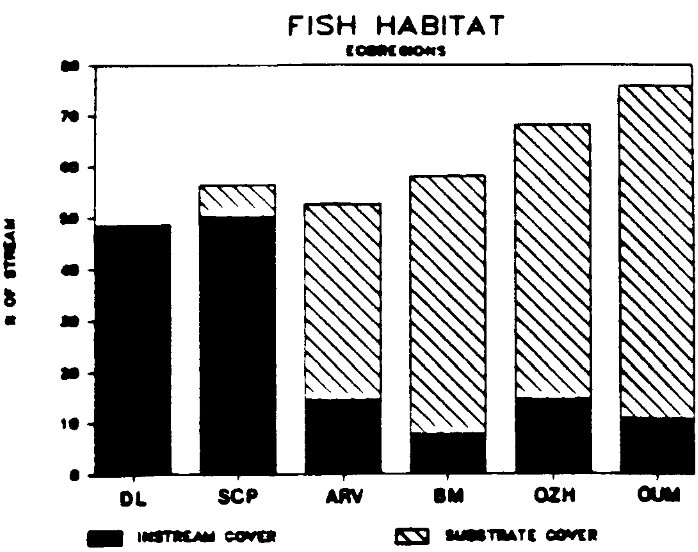
Value	Criteria	Descriptor
4	Abundant	Species or age group collected easily in a variety of habitats where species expected; numerous individuals seen with consideration of sampling gear limitations and typical abundance of such species; a dominant species of the species group.
3.5	Common to Abundant	
3	Common	Species or age groups collected in most areas where such species would exist; individuals frequently seen and apparently well established in the population; one of the more frequent species of the species group.
2.5	Present to Common	
2	Present	Species or age groups collected with enough frequency to indicate the likely presence of an established population but definitely a subordinate species in the species group.
1.5	Rare to Present	
1	Rare	Species or age groups represented by only one or very few individuals in the population; more than likely a remnant, migrant or a displaced species.

Values are assigned to the adult, intermediate and young age groups of each species; therefore, the maximum value for a species is 12 and the minimum is 1.

Table 2. List of reference streams within each ecoregion with watershed size, stream gradient and flows at sample sites.

Stream	Watershed Size (km^2)	Stream Gradient (m/km)	Summer Flow (m^3/s)	Spring Flow (m^3/s)
Ozark Highlands				
South Fork Spavinaw	46.8	4.8	0.04	0.51
Flint Creek	49.4	3.7	0.14	0.81
Yocum Creek	143.0	3.4	0.16	4.86
Long Creek	478.4	1.3	0.29	5.49
War Eagle Creek	683.8	0.8	0.75	3.06
Kings River	1367.6	0.9	1.46	7.56
Boston Mountains				
Indian Creek	122.2	6.1	T	0.57
Hurricane Creek	130.0	6.3	T	0.90
Archey Creek	278.2	2.7	0.02	3.66
Illinois Bayou	325.0	2.4	0.03	4.41
Lee Creek	436.8	2.9	0.11	9.00*
Mulberry River	969.8	2.6	0.19	9.00*
Arkansas River Valley				
Mill Creek	44.2	2.6	0	0.30
N. Cadron Creek	54.6	1.9	T	0.30
Ten Mile Creek	127.4	1.5	T	3.15
Dutch Creek	286.0	0.7	0.02	2.10
Petit Jean River	626.6	0.7	0.09	9.00*
Cadron Creek	800.8	0.1	0.45	15.00*
Ouachita Mountains				
Board Camp Creek	49.4	5.3	0.08	0
Little Missouri River	78.0	5.5	0.12	0
S Fork Ouachita	119.6	1.3	0.20	1
Cossatot River	312.0	7.6	0.52	2
Caddo River	756.6	2.5	4.02	15
Saline River	938.6	0.8	1.59	12
South Central Plains				
E Fork Tulip Creek	119.6	0.7	0.16	1
Cypress Creek	189.8	0.8	0.32	4
Whitewater Creek	59.8	0.5	0	0
Big Creek	153.4	0.5	0	0
Derrieusseaux Cr.	384.8	0.7	0	6
Freeo Creek	405.6	0.6	0	0
Hudgins Creek	486.2	0.3	0	9
L'Aigle Creek	603.2	0.5	0	5
Moro Creek	1172.6	0.3	0	10
Delta				
Boat Gunwale Slash	59.8	0.1	0.09	6
Second Creek	156.0	0.2	0.23	4
Village Creek	504.4	0.1	4.01	1
Bayou DeView	1196.0	0.1	5.73	15

T = flow less than 0.01
* = flow estimated



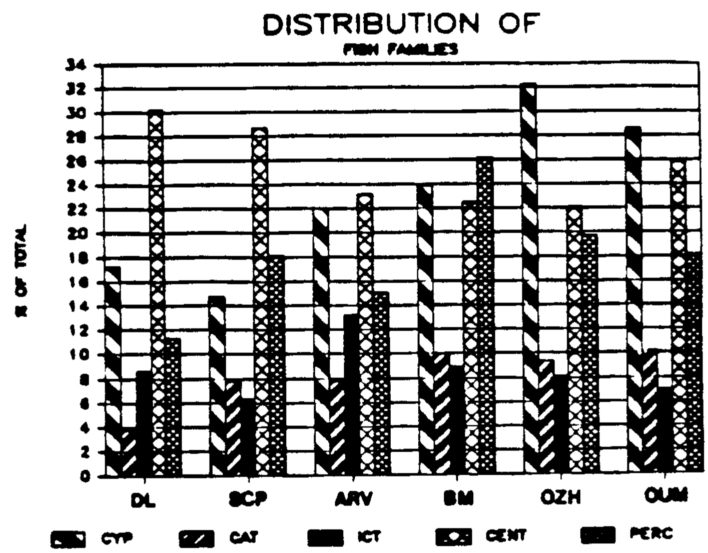


Fig. 2. Type of fish habitat in ecoregion reference streams as percent stream cross section and composed of either instream cover (brush, logs, debris) or substrate types which offer valuable fish cover (rubble, boulder, large boulder).

Fig. 3. Distribution of fishes within the families Cyprinidae (CYP), Catostomidae (CAT), Ictaluridae (ICT), Centrarchidae (CENT) and Percidae (PERC) for reference streams within each ecoregion.

which may include changes in turbidity, water temperature, dissolved solids, organic enrichment, lowered dissolved oxygen, as well as, physical habitat modification (i.e. substrate disruption, cover removal, water level fluctuations and flow modifications). There was a surprising consistency among the lists supplied.

Results and Discussion

General location of each sample site on the selected reference streams within Arkansas' six ecoregions are depicted in Figure 1. A list of the reference streams with the size of the watershed and stream gradient at the sample site are provided in Table 2. Also included are the stream flows which existed during the spring and summer sample periods. The range of watershed sizes among all sites was from 44.2 to 1367.6 km^2. Stream gradients were from 0.095 to 7.6 m/km.

Fish habitat was measured at each site during the summer sampling along numerous stream transects. Instream fish cover such as brush, logs, debris, undercut banks, aquatic vegetation and low-overhanging vegetation was measured directly along each transect and converted to percent of stream width. Stream substrate was also measured along each transect; however, a value relative to the value of different substrates as fish cover was applied to the percent of each substrate type. These values are as

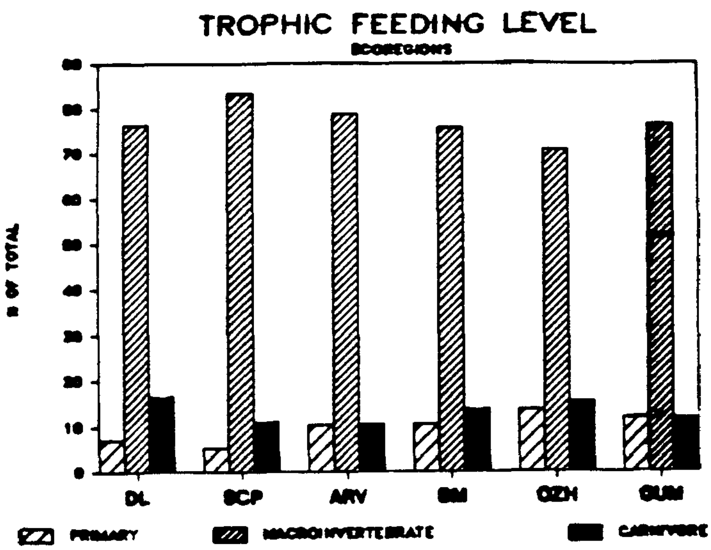
Table 3. Lists of key and indicator fish species within each of Arkansas's ecoregions.

Key Species	Ecoregion	Indicator Species
	Ozark Highlands	
Duskystripe shiner		Banded sculpin
Northern hogsucker		Ozark madtom
Slender madtom		Southern redbelly dace
"Rock basses"		Whitetail shiner
Orangethroat darter		Ozark minnow
Rainbow darter		
Smallmouth bass		
	Boston Mountains	
Bigeye shiner		Shadow bass
Black redhorse		Wedgespot shiner
Slender madtom		Longnose darter
Longear sunfish		Fantail darter
Greenside darter		
Smallmouth bass		
	Arkansas River Valley	
Bluntnose minnow		Orangespotted sunfish
Golden redhorse		Blackside darter
Yellow bullhead		Madtoms
Longear sunfish		
Redfin darter		
Spotted bass		
	Ouachita Mountains	
Bigeye shiner		Shadow bass
Northern hogsucker		Gravel chub
Freckled madtom		Northern studfish
Longear sunfish		Striped shiner
Orangebelly darter		
Smallmouth bass		
	Typical South Central Plains	
Redfin shiner		Pirate perch
Spotted sucker		Warmouth
Yellow bullhead		Spotted sunfish
Flier		Dusky darter
Slough darter		Creek chubsucker
Grass pickerel		Banded pygmy sunfish
	Springwater-Influenced South Central Plains	
Redfin shiner		Pirate perch
Blacktail redhorse		Golden redhorse
Freckled madtom		Spotted bass
Longear sunfish		Scaly sand darter
Creole darter		Striped shiner
Grass pickerel		Banded pygmy sunfish
	Least-Altered Delta	
Ribbon shiner		Mosquitofish
Smallmouth buffalo		Pugnose minnow
Yellow bullhead		Pirate perch
Bluegill		Tadpole madtom
Bluntnose darter		Banded pygmy sunfish
Largemouth bass		
	Channel-Altered Delta	
Blacktail shiner		Mosquitofish
Freshwater drum		Gizzard shad
Carp		Emerald shiner
Channel catfish		
Green sunfish		
Spotted gar		

follows: mud/silt, sand and bedrock = 0; gravel = 0.5; rubble, boulder and large boulder = 1. The different fish habitat within the ecoregions are demonstrated in Figure 2. Both the Delta and South Central Plains ecoregions are dominated by instream fish habitat such as brush, logs and debris. The Arkansas River Valley is highly variable in the type of fish habitat; however, from all sample sites, approximately 30% of the fish habitat is similar to that of the Delta and South Central plains region and about 70% is dominated by substrate types which provide desirable fish cover. The Boston Mountains, Ozark Highlands and Ouachita Mountains ecoregion streams are heavily dominated by fish habitat provided by substrate. These differences in fish habitat among the ecoregions produce distinctly different fish communities.

The distribution of fishes within the five major fish families of the State are shown for each ecoregion in Figure 3. The Delta and South Central Plains ecoregions are distinctly dominated by the Centrarchidae. The Arkansas River Valley is also dominated by Centrarchidae but Cyprinidae is only slightly sub-dominant. Percidae dominates the Boston Mountains fishes but are followed closely by Cyprinidae and Centrarchidae. The Ozark Highlands are strongly dominated by Cyprinidae followed by Centrarchidae and Percidae. Similarly the Ouachita Mountains communities are dominated by Cyprinidae although not as distinctly as in the Ozark Highlands.

The secondary trophic feeding level (macroinvertebrate feeding fishes) dominates the fish



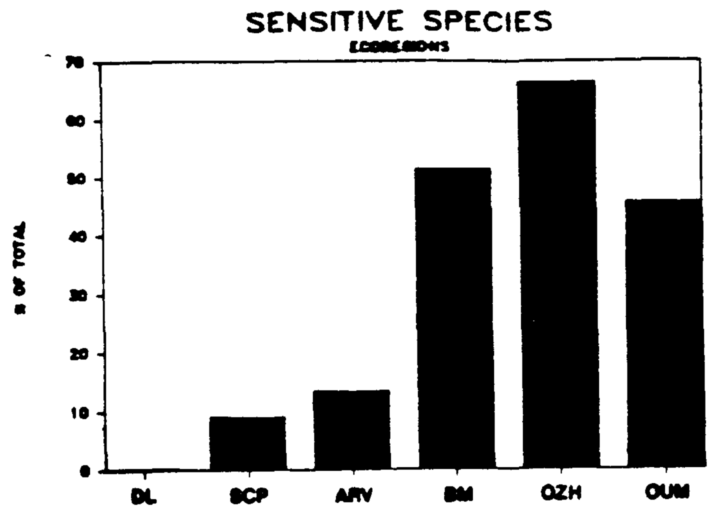


Fig. 4. Distribution of fishes within the trophic feeding levels of primary feeders, macroinvertebrate feeders and carnivores for reference streams within each ecoregion.

Fig. 5. Composition of sensitive fish species from reference streams within each ecoregion.

community of all regions (Fig. 4). Primary feeding fishes are least abundant in the South Central Plains ecoregion where two samples contained no primary feeders. They are most abundant in the Ozark Highlands. This region also contains the highest levels of nitrogen in the water of reference streams.

Sensitive fish species make up less than 0.2% of the relative abundance value of Delta ecoregion communities (Fig. 5). South Central Plains and Arkansas River Valley fish communities contain approximately 10 to 15% sensitive species. In contrast, sensitive species make up about 50% or more of the communities in the Ozark Highlands, Boston Mountains and Ouachita Mountains ecoregions. Over 66% of the Ozark Highlands fishes are sensitive species.

The average number of species collected per site was very similar among the ecoregions. However, the total number of species collected per ecoregion showed some variation (Fig. 6). The greatest number of species was collected from Arkansas River Valley streams followed by South Central Plains streams. The Delta ecoregion was lowest in species richness. Although it is realized that not all species present within each ecoregion were collected, it is felt that the majority of the more common species within the least-disturbed streams were identified. Areas inadequately sampled within the ecoregions were the large rivers.

All species collected within each ecoregion by sample site are listed in Appendix 1. The relative abundance value for each species is given for all sites where the species was collected. The species are listed in descending order of abundance within all reference streams of the ecoregion. From this data, a list of key species was

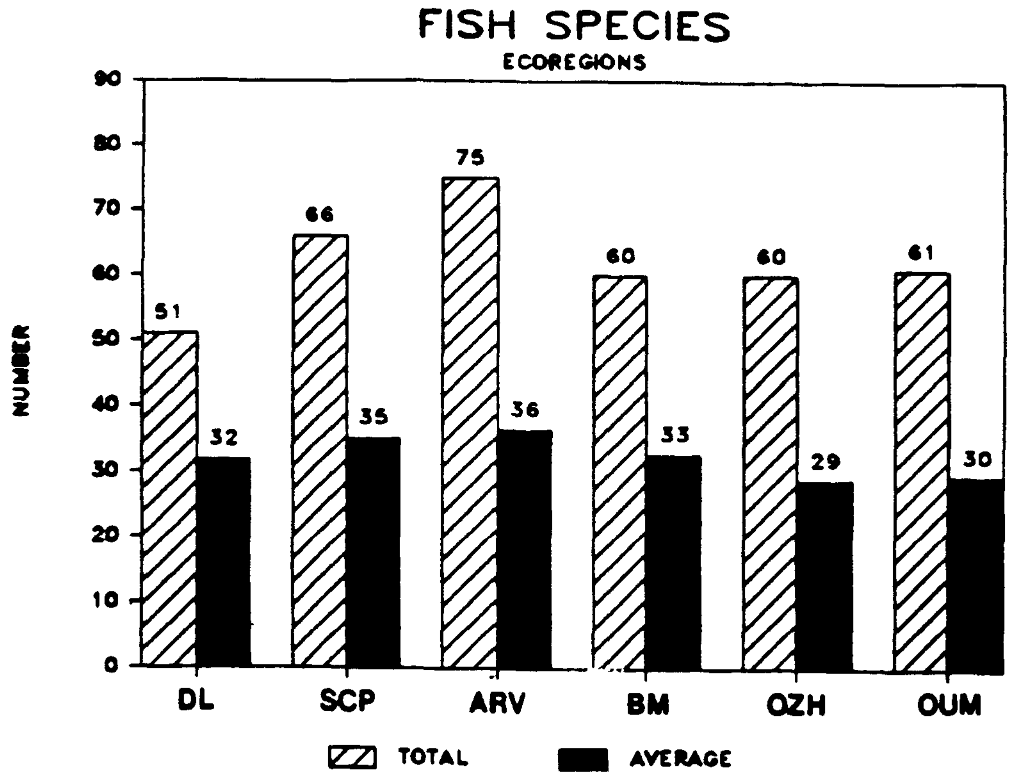


Fig. 6. Average and total number of fish species collected from all reference streams within each ecoregion.

Table 4. Similarity indices from comparisons of relative abundance values of the ten most abundant fish species of all ecoregions.

	Ecoregions				
	Boston Mountains	Ozark Highlands	Arkansas River Valley	Delta	South Central Plains
Ouachita Mountains	62	32	21	11	11
Boston Mountains		39	40	10	10
Ozark Highlands			19	9	9
Arkansas River Valley				36	29
Delta					58

developed for each ecoregion. This list represents the species which are dominant in most or all of the samples from the ecoregion within the five major fish families (i.e. Cyprinidae, Catostomidae, Ictaluridae, Centrarchidae and Percidae) and the dominant predator species. Also, a list of indicator species was developed which may or may not be numerically dominant and may not occur exclusively within one ecoregion, but their presence more than likely indicates the ecosystem from which they were collected.

In the South Central Plains and Delta ecoregions, different fish communities were found in a notable habitat variation within each ecoregion. A few drainage systems within the South Central Plains ecoregion are substantially influenced by springwater discharges. Such systems are represented by the East Fork Tulip and Cypress Creek reference streams. For these systems a different group of key and indicator species were identified. Also, within the Delta Ecoregion, a large majority of streams were channelized to facilitate drainage of agricultural lands. This severe physical alteration of the fish habitat has significantly changed the fish community. From a satellite project of the reference stream study and from numerous use attainability analyses performed on Delta ecoregion streams, key and indicator species were developed for the channel-altered Delta streams. A list of key and indicator species for the distinctive ecoregion fish communities is in Table 3.

A similarity index, modified from Odum (1971), was used to compare the 10 most abundant species within each ecoregion. These groups of fish included almost all key species and several indicator species from each ecoregion. Odum's index compares the number of species common in two populations with the total number of species from each population. This index was modified to use relative abundance values of the species as follows:

$$SI = \frac{C}{A + B + D} \times 100$$

Where, SI = similarity index (range from 0 to 100; 100 = identical populations);
A = total relative abundance value of sample A;
B = total relative abundance value of sample B;
C = sum of relative abundance values of species common to both samples;
D = sum of difference in relative abundance values of species common to both samples.

All possible comparisons among the six ecoregions were made (Table 4). The greatest similarity exist between the Ouachita Mountains and the Boston Mountain fishes. The least similarity is between the Ozark Highlands versus South Central Plains and between the Ozark Highlands versus Delta fishes. It is apparent from the similarity indicies that there is very little similarity of the 10 most abundant fishes from each of six ecoregions within the State. This substantiates the distinctiveness of these ecoregions as reflected in the fish communities of the least-disturbed streams and supports the use of fish communities as designated uses of a waterbody. Such characteristics may also be used as an indicator of environmental impacts.

Acknowledgements

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Appendix Table 1. List of fish species with relative abundance values at each reference stream where species collected within the Ozark Highlands, Boston Mountains, Arkansas River Valley, Ouachita Mountains, and South Central Plains and Delta Ecoregions.

Ozark Highlands Ecoregion

FISH	SPECIES	SPAV.CR.	FLINT CR	YOCUM CR	LONG CR	WAR EGLE	KINGS	SUM
*(Notropis pilsbryi)	Duskystripe shiner	12.0	12.0	12.0	12.0	12.0	10.5	70.5
(Campostoma anomalum)	Stoneroller	10.5	12.0	12.0	12.0	12.0	12.0	70.5
*(Hypentelium nigricans)	Northern hogsucker	2.0	9.0	12.0	12.0	10.5	9.0	54.5
*(Cottus carolinae)	Banded sculpin	12.0	10.5	12.0	9.0	7.5		51.0
*(Etheostoma caeruleum)	Rainbow darter			10.5	12.0	12.0	12.0	46.5
(Lepomis megalotis)	Longear		9.0	8.0	9.0	10.5	9.0	45.5
*(Noturus exilis)	Slender madtom	12.0	10.5	9.0	6.0	3.0		40.5
*(Ambloplites constellatus)	Ozark bass			12.0	9.5	9.0	9.0	39.5
*(Micropterus dolomieui)	Smallmouth bass	2.0	10.5	8.5	4.0	1.0	10.5	36.5
(Notropis nubilus)	Ozark minnow		6.0	1.0	10.5	12.0	7.0	36.5
(Micropterus punctulatus)	Spotted bass				12.0	12.0	12.0	36.0
*(Noturus albater)	Ozark madtom			4.0	12.0	9.0	8.0	33.0
*(Etheostoma juliae)	Yoke darter				12.0	12.0	9.0	33.0
(Lepomis cyanellus)	Green sunfish	1.5	7.5	6.0	4.0	5.0	7.0	31.0
*(Moxostoma duquesnei)	Black redhorse				10.5	8.0	12.0	30.5
(Percina caprodes)	Logperch			1.0	12.0	2.0	12.0	27.0
(Etheostoma spectabile)	Orangethroat darter	9.0	12.0	6.0	S			27.0
*(Etheostoma zonale)	Banded darter				9.0	4.5	12.0	25.5
*(Notropis rubellus)	Rosyface shiner	S	1.5		6.0	9.0	7.5	24.0
*(Etheostoma flabellare)	Fantail darter	12.0		12.0				24.0
*(Etheostoma blennioides)	Greenside darter			S	9.0	6.0	9.0	24.0
*(Phoxinus erythrogaster)	Southern redbelly dace	12.0	9.0	S				21.0
*(Nocomis asper)	Redspot chub	9.0	12.0					21.0
*(Ambloplites rupestris)	Rock bass	9.0	12.0					21.0
*(Nocomis biguttatus)	Hornyhead chub			10.5	9.0			19.5
(Lepomis macrochirus)	Bluegill		1.0	6.0	3.0	2.0	6.5	18.5
(Moxostoma erythrurum)	Golden redhorse				1.0	8.0	8.0	17.0
*(Semotilus atromaculatus)	Creek chub	10.5	6.0					16.5
(Fundulus olivaceus)	Blackspotted topminnow		6.0	4.0		4.0	1.0	15.0
*(Moxostoma carinatum)	River Redhorse				4.0	1.0	9.0	14.0
*(Hybopsis dissimilis)	Streamline chub				6.0		8.0	14.0
*(Fundulus catenatus)	Northern studfish		4.5	8.0	1.0			13.5
(Notropis chrysocephalus)	Striped shiner				6.0	5.5	2.0	13.5
*(Notropis boops)	Bigeye shiner				4.5	4.0	4.0	12.5
(Ictalurus punctatus)	Channel catfish				4.5	S	8.0	12.5
*(Etheostoma euzonum)	Arkansas saddled darter						12.0	12.0
(Gambusia affinis)	Mosquitofish		10.5					10.5
(Pimephales notatus)	Bluntnose minnow				7.5	1.0	1.0	9.5
*(Etheostoma punctulatum)	Stippled darter	S	9.0					9.0
*(Notropis galacturus)	Whitetail shiner				6.0	1.0	1.0	8.0
(Dorosoma cepedianum)	Gizzard shad				1.5		6.0	7.5
(Micropterus salmoides)	Largemouth bass			1.0	1.0	1.0	4.0	7.0
*(Notropis whipplei)	Steelcolor shiner					4.0	1.0	5.0
(Pylodictis olivaris)	Flathead catfish					S	4.0	4.0
(Cyprinus carpio)	Carp				1.0	S	2.0	3.0
*(Notropis greenei)	Wedgespot shiner						2.0	2.0
*(Hybopsis amblops)	Bigeye chub						2.0	2.0
*(Etheostoma stigmaeum)	Speckled darter					2.0		2.0
(Labidesthes sicculus)	Brook silversides				1.0	1.0		2.0
*(Noturus flavater)	Checkered madtom					1.0		1.0
*(Notropis telescopus)	Telescope shiner					1.0		1.0
(Lepomis hybrid)	Hybrid sunfish	1.0						1.0
(Catostomus commersoni)	White sucker		1.0					1.0
*(Stizostedion vitreum)	Walleye						S	0.0
(Lepisosteus osseus)	Longnose gar					S		0.0
(Lepisosteus oculatus)	Spotted gar						S	0.0
(Ictalurus natalis)	Yellow bullhead				S			0.0
(Ictalurus melas)	Black bullhead						S	0.0
(Carpiodes velifer)	Highfin carpsucker					S	S	0.0
(Carpiodes cyprinus)	Quillback carpsucker					S		0.0
	NUMBER OF SPECIES=	16	21	22	36	39	39	60.0

* - SENSITIVE SPECIES
S - SPRING COLLECTION ONLY

Boston Mountains Ecoregion

FISH	SPECIES	INDIAN	MURR	ARCHEY	ILL. BY.	LEE CR.	MULBRY	SUM
(Campostoma anomalum)	Stoneroller	12.0	12.0	9.0	12.0	12.0	12.0	69.0
*(Notropis boops)	Bigeye shiner	12.0	12.0	9.0	10.5	12.0	12.0	67.5
(Lepomis megalotis)	Longear	11.0	12.0	10.5	12.0	10.0	12.0	67.5
*(Noturus exilis)	Slender madtom	10.5	10.5	12.0	10.5	10.5	12.0	66.0
*(Etheostoma blennioides)	Greenside darter	10.5	12.0	12.0	9.0	7.5	9.0	60.0
(Lepomis cyanellus)	Green sunfish	8.0	10.5	10.0	12.0	6.0	10.5	57.0
*(Micropterus dolomieui)	Smallmouth bass	9.0	9.0	8.5	11.5	10.5	4.0	52.5
(Micropterus punctulatus)	Spotted bass	7.0	6.5	9.0	10.5	9.5	9.0	51.5
*(Etheostoma zonale)	Banded darter	7.5	10.5	7.5	9.0	6.0	9.0	49.5
*(Moxostoma duquesnei)	Black redhorse	11.0	9.0	10.0	10.5	7.5		48.0
(Pimephales notatus)	Bluntnose minnow	9.0	1.0	10.5	9.0	5.0	9.0	43.5
(Labidesthes sicculus)	Brook silversides	12.0	6.0	9.0	10.5	4.0	2.0	43.5
(Fundulus olivaceus)	Blackspotted topminnow	10.5	8.0	9.0	6.0	4.0	5.0	42.5
*(Hypentelium nigricans)	Northern hogsucker	9.0	6.0	6.5	7.5	4.0	8.5	41.5
*(Notropis greenei)	Wedgespot shiner		6.0	9.0	9.0	7.5	9.0	40.5
(Etheostoma spectabile)	Orangethroat darter	12.0	10.5		12.0	1.0	S	35.5
*(Etheostoma flabellare)	Fantail darter	4.5	6.0			12.0	12.0	34.5
(Etheostoma whipplei)	Redfin darter	12.0		6.0	12.0	2.0	2.0	34.0
*(Notropis whipplei)	Steelcolor shiner	2.5	1.5	7.0	9.0	6.0	7.5	33.5
(Moxostoma erythrurum)	Golden redhorse	9.0	4.5	6.0	1.0	9.0	3.0	32.5
*(Percina nasuta)	Longnose darter			6.5	7.5	5.0	6.0	25.0
*(Etheostoma punctulatum)	Stippled darter	4.5		3.0	12.0	4.0		23.5
(Percina caprodes)	Logperch	1.5		9.5		4.0	7.5	22.5
(Ictalurus punctatus)	Channel catfish	9.0	3.0	1.0		S	7.0	20.0
*(Ambloplites ariommus)	Shadow bass		1.0	10.5	6.0			17.5
(Micropterus salmoides)	Largemouth bass				2.5	6.0	6.0	14.5
(Lepomis macrochirus)	Bluegill	1.0		3.5	2.0	5.0	2.0	13.5
*(Notropis pilsbryi)	Duskystripe shiner					12.0		12.0
*(Etheostoma caeruleum)	Rainbow darter			12.0				12.0
*(Noturus albater)	Ozark madtom			9.0				9.0
*(Hybopsis dissimilis)	Streamline chub			9.0				9.0
*(Etheostoma moorei)	Yellowcheek darter			9.0				9.0
(Notropis nubilus)	Ozark minnow					9.0		9.0
*(Etheostoma stigmaeum)	Speckled darter			7.5		1.0		8.5
(Ictalurus natalis)	Yellow bullhead		S	2.0	6.5			8.5
*(Etheostoma euzonum)	Arkansas saddled darter			8.0				8.0
(Pylodictis olivaris)	Flathead catfish			6.0	1.0		1.0	8.0
*(Pimephales tenellus)	Slim minnow			6.0				6.0
*(Percina maculata)	Blackside darter	1.0	S			2.5	2.0	5.5
(Esox americanus)	Grass pickerel	1.0			4.5			5.5
(Cyprinus carpio)	Carp	1.5				2.0	2.0	5.5
(Aplodinotus grunniens)	Freshwater drum	2.0		2.0			1.0	5.0
*(Fundulus catenatus)	Northern studfish					4.0		4.0
(Fundulus notatus)	Blackstripe topminnow			2.0		1.0		3.0
(Lepomis gulosus)	Warmouth			1.0			1.0	2.0
*(Semotilus atromaculatus)	Creek chub			1.0				1.0
*(Percina copelandi)	Channel darter						1.0	1.0
(Noturus miurus)	Brindled madtom			1.0				1.0
(Morone chrysops)	White bass			1.0				1.0
(Lepomis hybrid)	Hybrid sunfish				1.0			1.0
(Lepomis humilis)	Orangespotted sunfish					1.0		1.0
(Lepisosteus osseus)	Longnose gar					1.0	S	1.0
(Ichthyomyzon sp.)	Lamprey larvae		S	1.0				1.0
(Dorosoma cepedianum)	Gizzard shad			1.0	S			1.0
*(Stizostedion vitreum)	Walleye			S				0.0
*(Moxostoma carinatum)	River Redhorse				S	S	S	0.0
(Notropis emiliae)	Pugnose minnow		S					0.0
(Lepisosteus oculatus)	Spotted gar			S		S		0.0
(Ictiobus bubalus)	Smallmouth buffalo						S	0.0
(Carpiodes carpio)	River carpsucker						S	0.0
	NUMBER OF SPECIES=	27	25	43	30	37	34	60.0

Arkansas River Valley Ecoregion

FISH	SPECIES	MILL CR.	N.FK.CAD	TEN MI.	DUTCH	PET JEAN	CADRON	SUM
(Lepomis megalotis)	Longear	10.5	10.5	12.0	12.0	12.0	6.0	63.0
(Pimephales notatus)	Bluntnose minnow	12.0	9.0	12.0	12.0	12.0	4.0	61.0
(Etheostoma whipplei)	Redfin darter	12.0	12.0	9.0	12.0	12.0		57.0
(Fundulus olivaceus)	Blackspotted topminnow	9.0	12.0	9.0	9.0	9.0	7.5	55.5
(Lepomis cyanellus)	Green sunfish	12.0	10.5	7.5	10.5	9.0	2.0	51.5
(Micropterus punctulatus)	Spotted bass	9.0	7.5	7.5	8.0	9.0	8.0	49.0
(Ictalurus natalis)	Yellow bullhead	10.5	12.0	12.0	12.0	2.0		48.5
(Lepomis macrochirus)	Bluegill	9.0	6.0	7.5	7.5	6.0	12.0	48.0
*(Noturus exilis)	Slender madtom	9.0	12.0	9.0	5.5	12.0		47.5
(Labidesthes sicculus)	Brook silversides	9.0	4.5	9.0	9.0	9.0	6.0	46.5
(Campostoma anomalum)	Stoneroller	S	12.0	6.0	10.5	9.0		37.5
(Notropis umbratilis)	Redfin shiner	9.0	9.0	6.0	4.0	6.0	2.0	36.0
(Moxostoma erythrurum)	Golden redhorse	10.5	9.0	2.0	2.0	9.0	1.0	33.5
*(Notropis boops)	Bigeye shiner	9.0	9.0	9.0	6.0			33.0
(Erimyzon oblongus)	Creek chubsucker	5.0	12.0	9.0	2.0			28.0
(Esox americanus)	Grass pickerel	9.0	7.5		10.0	1.0		27.5
(Etheostoma spectabile)	Orangethroat darter		9.0		10.5	7.5		27.0
(Minytrema melanops)	Spotted sucker	9.0	7.5	2.0	2.0	2.0	4.0	26.5
(Dorosoma cepedianum)	Gizzard shad	10.5				4.0	12.0	26.5
(Notropis emiliae)	Pugnose minnow	S		1.0	9.0	6.0	9.0	25.0
(Noturus gyrinus)	Tadpole madtom			7.5	9.0		7.5	24.0
(Micropterus salmoides)	Largemouth bass	1.5	7.5	8.0	1.0	2.0	4.0	24.0
(Aphredoderus sayanus)	Pirate perch	1.5	4.0	9.0	6.0		2.0	22.5
(Noturus miurus)	Brindled madtom				9.0	12.0		21.0
(Notropis fumeus)	Ribbon shiner				9.0	7.5	4.5	21.0
(Fundulus notatus)	Blackstripe topminnow			9.0	12.0			21.0
(Aplodinotus grunniens)	Freshwater drum	3.0				6.0	12.0	21.0
(Lepomis gulosus)	Warmouth	4.0		4.0	5.0	1.0	6.0	20.0
(Pimephales vigilax)	Bullhead minnow	S				7.5	12.0	19.5
(Percina caprodes)	Logperch	4.0	1.0		7.0	7.0		19.0
*(Notropis whipplei)	Steelcolor shiner		1.0		8.5	4.5	4.0	18.0
(Ictalurus punctatus)	Channel catfish				4.0	4.0	9.0	17.0
(Noturus nocturnus)	Freckled madtom				7.5	9.0		16.5
*(Percina sciera)	Dusky darter				6.0	6.0	4.0	16.0
(Lepomis humilis)	Orangespotted sunfish					9.0	6.0	15.0
(Lepomis punctatus)	Spotted sunfish	1.0	6.0		6.0		1.0	14.0
*(Etheostoma stigmaeum)	Speckled darter				6.0	7.5		13.5
(Gambusia affinis)	Mosquitofish					9.0	4.0	13.0
(Notropis volucellus)	Mimic shiner						12.0	12.0
(Etheostoma gracile)	Slough darter				9.0		1.0	10.0
*(Percina copelandi)	Channel darter					7.5	2.0	9.5
*(Etheostoma punctulatum)	Stippled darter			9.0				9.0
(Notropis atherinoides)	Emerald shiner						9.0	9.0
*(Etheostoma caeruleum)	Rainbow darter			7.5				7.5
(Notropis chrysocephalus)	Striped shiner			7.5				7.5
(Etheostoma proeliare)	Cypress darter			7.5				7.5
*(Etheostoma flabellare)	Fantail darter		7.0					7.0
(Notropis venustus)	Blacktail shiner					1.0	6.0	7.0
*(Percina maculata)	Blackside darter	3.0	1.5	1.0	1.0			6.5
*(Etheostoma blennioides)	Greenside darter			6.0				6.0
(Etheostoma chlorosomum)	Bluntnose darter			6.0				6.0
(Pomoxis annularis)	White crappie				1.0		4.5	5.5
(Ictiobus bubalus)	Smallmouth buffalo					4.0	1.0	5.0
(Lepomis microlophus)	Redear	4.0						4.0
(Elassoma zonatum)	Banded pygmy sunfish						4.0	4.0
(Amia calva)	Bowfin	3.0					1.0	4.0
(Notemigonus crysoleucas)	Golden shiner		1.0	1.5	1.0			3.5
*(Hypentelium nigricans)	Northern hogsucker	1.0	1.0	1.0				3.0
(Pylodictis olivaris)	Flathead catfish					2.0	1.0	3.0
(Morone chrysops)	White bass					2.0	1.0	3.0
(Lepomis hybrid)	Hybrid sunfish			1.0	2.0			3.0
(Etheostoma asprigene)	Mud darter						3.0	3.0
(Lepisosteus oculatus)	Spotted gar				1.5	1.0		2.5
(Carpiodes carpio)	River carpsucker				S		2.5	2.5
(Ictalurus melas)	Black bullhead	S		2.0				2.0
(Esox niger)	Chain pickerel	1.0		1.0				2.0
(Pomoxis nigromaculatus)	Black crappie	1.5						1.5
*(Semotilus atromaculatus)	Creek chub			1.0				1.0
*(Etheostoma histrio)	Harlequin darter					1.0		1.0
(Ichthyomyzon sp.)	Lamprey larvae				1.0			1.0
*(Pimephales tenellus)	Slim minnow				S			0.0
*(Moxostoma carinatum)	River Redhorse					S		0.0
(Moxostoma macrolepidotum)	Shorthead redhorse				S			0.0
(Lepisosteus osseus)	Longnose gar					S		0.0
(Cyprinus carpio)	Carp				S			0.0
	NUMBER OF SPECIES=	30	27	35	44	41	38	75.0

Ouachita Mountains Ecoregion

·.SH SPECIES		WRD.CAMP	L.MO.	S.FK.OUA	COSSAT.	CADDO	SALINE	SUM
(Campostoma anomalum)	Stoneroller	12.0	12.0	12.0	12.0	12.0	12.0	72.0
(Lepomis megalotis)	Longear	10.5	12.0	12.0	12.0	12.0	9.5	68.0
*(Notropis boops)	Bigeye shiner	9.5	12.0	12.0	7.0	12.0	10.0	62.5
*(Etheostoma radiosum)	Orangebelly darter	12.0	12.0	12.0	12.0	12.0		60.0
*(Etheostoma blennioides)	Greenside darter	6.5	12.0	10.5		10.5	11.0	50.5
*(Micropterus dolomieui)	Smallmouth bass	4.0	9.5	10.5	8.0	6.0	7.0	45.0
(Noturus nocturnus)	Freckled madtom	9.0	2.0	9.0		10.5	12.0	42.5
(Lepomis cyanellus)	Green sunfish	9.0	6.0	9.0	6.0	9.0	2.0	41.0
(Notropis chrysocephalus)	Striped shiner	7.0	9.0	12.0		6.0	6.0	40.0
*(Fundulus catenatus)	Northern studfish	6.5	6.0	9.0	8.0	9.0		38.5
*(Hypentelium nigricans)	Northern hogsucker	7.0	9.0	9.0		5.0	6.5	36.5
(Pimephales notatus)	Bluntnose minnow	S	12.0	10.5		7.5	6.0	36.0
(Moxostoma erythrurum)	Golden redhorse	S		9.0	7.0	7.0	7.5	30.5
*(Etheostoma zonale)	Banded darter		1.0	7.5		9.0	12.0	29.5
(Percina caprodes)	Logperch		6.0	10.5	S	11.0	1.0	28.5
(Micropterus punctulatus)	Spotted bass			10.5		9.0	9.0	28.5
(Fundulus olivaceus)	Blackspotted topminnow	1.0	6.0	7.5	6.0	6.0	2.0	28.5
*(Moxostoma duquesnei)	Black redhorse		S	8.5	7.0	4.0	7.5	27.0
*(Ambloplites ariommus)	Shadow bass		7.5	8.0	2.0	4.5	5.0	27.0
*(Hybopsis x-punctata)	Gravel Chub					10.5	12.0	22.5
(Lepomis macrochirus)	Bluegill	4.0	1.0	6.0		7.0	2.0	20.0
*(Notropis whipplei)	Steelcolor shiner			7.5	1.0	6.0	4.0	18.5
(Ictalurus natalis)	Yellow bullhead	2.0	6.0	5.5	2.0			15.5
(Micropterus salmoides)	Largemouth bass		1.0	6.5	1.0	6.0		14.5
(Etheostoma whipplei)	Redfin darter						12.0	12.0
(Labidesthes sicculus)	Brook silversides	1.0	7.5			1.0	2.0	11.5
*(Nocomis asper)	Redspot chub			9.0				9.0
(Minvtrema melanops)	Spotted sucker			6.0		3.0		9.0
(Lepomis microlophus)	Redear			6.0		2.0	1.0	9.0
*(Notropis snelsoni)	Ouachita Mt. shiner				8.5			8.5
(Esox americanus)	Grass pickerel			6.0			2.0	8.0
(Notropis umbratilis)	Redfin shiner		6.0	S			1.0	7.0
(Dorosoma cepedianum)	Gizzard shad					2.0	4.5	6.5
*(Noturus eleutherus)	Mountain madtom					6.0		6.0
*(Etheostoma collettei)	Creole darter						5.0	5.0
(Notropis atherinoides)	Emerald shiner			5.0				5.0
(Lepomis gulosus)	Warmouth			S		4.0	1.0	5.0
(Fundulus notatus)	Blackstripe topminnow					4.5	S	4.5
*(Noturus taylori)	Caddo madtom			1.0		2.0	-	3.0
(Ichthyomyzon sp.)	Lamprey larvae			2.0		1.0		3.0
(Lepomis hybrid)	Hybrid sunfish	1.5				1.0		2.5
*(Pimephales tenellus)	Slim minnow	2.0						2.0
(Pylodictis olivaris)	Flathead catfish				2.0			2.0
(Noturus miurus)	Brindled madtom						2.0	2.0
(Notropis fumeus)	Ribbon shiner		1.0			1.0		2.0
(Erimyzon oblongus)	Creek chubsucker	1.0		S		1.0		2.0
*(Semotilus atromaculatus)	Creek chub		1.0					1.0
*(Percina copelandi)	Channel darter						1.0	1.0
*(Etheostoma histrio)	Harlequin darter						1.0	1.0
(Lepomis punctatus)	Spotted sunfish						1.0	1.0
(Ichthyomyzon castaneus)	Chestnut lamprey						1.0	1.0
(Etheostoma chlorosomum)	Bluntnose darter			1.0				1.0
(Aphredoderus sayanus)	Pirate perch			S		1.0		1.0
*(Salmo gairdneri)	Rainbow trout		S					0.0
*(Noturus lachneri)	Ouachita madtom						S	0.0
*(Moxostoma carinatum)	River Redhorse					S	S	0.0
(Pomoxis nigromaculatus)	Black crappie					S		0.0
(Lepisosteus osseus)	Longnose gar				S	S		0.0
(Ictalurus punctatus)	Channel catfish					S	S	0.0
(Ictalurus melas)	Black bullhead	S						0.0
(Ichthyomyzon gagei)	Southern brook lamprey			S				0.0
	NUMBER OF SPECIES=	21	25	36	18	40	37	61.0

South Central Plains Ecoregion

FISH SPECIES		TULIP	CYPRS	W.WTR	BIG	DERSX	FREED	HDGNS	L'AGL	MORO	SUM
(Aphredoderus sayanus)	Pirate perch	9.0	10.5	10.0	12.0	9.0	12.0	12.0	9.0	10.5	94.0
(Lepomis gulosus)	Warmouth	6.5	9.0	12.0	7.0	12.0	7.0	10.0	10.5	10.0	84.0
(Lepomis megalotis)	Longear	12.0	12.0	9.0	5.5	10.5	10.0	6.0	6.0	6.5	77.5
(Fundulus olivaceus)	Blackspotted topminnow	7.5	9.0	7.5	6.0	9.0	10.5	11.0	9.0	8.0	77.5
(Centrarchus macropterus)	Flier	4.5	9.0	12.0	10.0	12.0	6.0	8.0	10.5	5.0	77.0
(Esox americanus)	Grass pickerel	9.0	9.0	9.0	9.0	9.0	9.0	3.0	12.0	7.0	76.0
(Minytrema melanops)	Spotted sucker	2.0	7.0	9.0	1.0	11.0	3.0	12.0	12.0	12.0	69.0
(Ictalurus natalis)	Yellow bullhead	6.0	12.0	1.0	2.0	12.0	6.0	8.0	12.0	9.0	68.0
(Gambusia affinis)	Mosquitofish	4.5	4.5	9.0	9.0	6.5	7.5	9.0	9.0	7.5	66.5
(Etheostoma gracile)	Slough darter	7.5	4.5	9.0	9.0	10.0	2.0	12.0	7.0	4.0	65.0
(Notropis umbratilis)	Redfin shiner	12.0	12.0	6.0	6.0	9.0	10.5	6.5		2.0	64.0
(Lepomis macrochirus)	Bluegill	4.5		4.0	5.5	7.5	3.0	9.0	12.0	9.0	54.5
(Lepomis cyanellus)	Green sunfish	6.5	9.0	6.0	6.5	5.0	4.0	4.0	6.0	4.0	51.0
(Etheostoma whipplei)	Redfin darter	6.0	9.0	4.0	2.0	6.0	7.0	3.0	6.0	7.5	50.5
(Elassoma zonatum)	Banded pygmy sunfish	9.0	6.0	9.0	9.0	4.0	4.0	2.0	1.0	4.0	48.0
*(Etheostoma collettei)	Creole darter	9.0	7.5	1.0	S	1.0	9.0	2.0	7.0	11.0	47.5
(Lepomis punctatus)	Spotted sunfish	5.0	6.0	5.0		5.0	9.0	3.0	9.0	4.5	46.5
*(Percina maculata)	Blackside darter	6.0	6.0		7.0	5.5	8.0	5.5		5.0	43.0
(Etheostoma chlorosomum)	Bluntnose darter		1.0	1.0	2.0	6.0	6.0	12.0	4.5	9.0	41.5
(Micropterus salmoides)	Largemouth bass	7.5	2.0	1.0		4.0	2.0	9.0	7.5	7.0	40.0
*(Percina sciera)	Dusky darter	6.5	8.0		1.0	4.0	7.0		1.0	9.0	36.5
(Fundulus notatus)	Blackstripe topminnow	9.0	7.5	4.0				4.0	6.0	6.0	36.5
(Amia calva)	Bowfin	3.0	7.0	3.0		6.0	1.0	7.0	2.5	6.5	36.0
(Esox niger)	Chain pickerel		10.5	S		4.0		4.5	10.0	6.0	35.0
(Notropis chrysocephalus)	Striped shiner	9.0	7.5		6.0	2.0	1.0	4.0		5.0	34.5
(Notropis emiliae)	Pugnose minnow	6.0	6.0		1.0	3.0	2.0	6.0	2.0	7.0	33.0
(Hybognathus nuchalus)	Silvery minnow	1.5			2.0	7.0		12.0		9.0	31.5
(Erimyzon oblongus)	Creek chubsucker	2.0		6.0	8.0	5.0	5.0	1.0	2.0	2.5	31.5
*(Moxostoma poecilurum)	Blacktail redhorse	12.0	7.5			7.0	2.0	S		2.0	30.5
(Hybognathus hayi)	Cypress minnow					6.0		9.0	4.0	9.0	28.0
(Notropis fumeus)	Ribbon shiner					6.0		2.0	6.0	12.0	26.0
(Noturus nocturnus)	Freckled madtom	10.5	12.0							3.0	25.5
(Etheostoma proeliare)	Cypress darter		1.0	9.0			6.0		4.5	4.0	24.5
(Noturus gyrinus)	Tadpole madtom		6.0			4.0	9.0	2.0	1.0	2.0	24.0
(Notemigonus crysoleucas)	Golden shiner		1.0		9.0	8.0			2.0	4.0	24.0
(Notropis texanus)	Weed shiner					4.0			2.0	12.0	18.0
(Micropterus punctulatus)	Spotted bass	6.0	7.5					2.0			15.5
(Moxostoma erythrurum)	Golden redhorse	6.5	6.0							1.0	13.5
(Pomoxis nigromaculatus)	Black crappie		1.0			1.0		2.0	2.0	7.0	13.0
*(Ammocrypta vivax)	Scaly sand darter	9.0	1.0							2.0	12.0
(Noturus miurus)	Brindled madtom	10.5									10.5
(Percina caprodes)	Logperch							1.0		9.0	10.0
(Labidesthes sicculus)	Brook silversides		1.5			2.0	1.0		1.0	4.0	9.5
(Notropis atherinoides)	Emerald shiner						4.5		4.0		8.5
(Pimephales notatus)	Bluntnose minnow	7.5									7.5
(Aplodinotus grunniens)	Freshwater drum									5.0	5.0
(Pimephales vigilax)	Bullhead minnow									4.0	4.0
(Ichthyomyzon gagei)	Southern brook lamprey	3.0	1.0								4.0
(Campostoma anomalum)	Stoneroller	1.0	3.0								4.0
(Lepomis symmetricus)	Bantam sunfish			3.0							3.0
(Lepomis hybrid)	Hybrid sunfish					1.0	1.0	1.0			3.0
(Anguilla rostrata)	American eel		3.0								3.0
*(Percina ouachitae)	Saddleback darter									2.5	2.5
*(Etheostoma stigmaeum)	Speckled darter	1.5	1.0								2.5
*(Etheostoma parvipinne)	Goldstripe darter		1.0				1.0				2.0
(Percina shumardi)	River darter									2.0	2.0
(Cyprinus carpio)	Carp									2.0	2.0
*(Notropis amnis)	Pallid shiner									1.0	1.0
*(Hypentelium nigricans)	Northern hogsucker		1.0								1.0
*(Fundulus catenatus)	Northern studfish		1.0								1.0
*(Ammocrypta asprella)	Crystal darter									1.0	1.0
(Notropis venustus)	Blacktail shiner									1.0	1.0
(Lepomis microlophus)	Redear							1.0			1.0
(Lepisosteus oculatus)	Spotted gar							1.0			1.0
(Pomoxis annularis)	White crappie									S	0.0
(Ichthyomyzon castaneus)	Chestnut lamprey		S								0.0
	NUMBER OF SPECIES=	36	43	25	24	36	32	37	33	50	66.0

Delta Ecoregion

FISH SPECIES		BOAT 6	SECOND	VILGE CR.	BY DeVIEW	SUM
(Gambusia affinis)	Mosquitofish	12.0	10.5	12.0	9.0	43.5
(Aphredoderus sayanus)	Pirate perch	10.0	12.0	12.0	8.0	42.0
(Lepomis macrochirus)	Bluegill	6.5	12.0	8.0	7.5	34.0
(Fundulus olivaceus)	Blackspotted topminnow	8.5	9.0	10.5	1.0	29.0
(Lepomis punctatus)	Spotted sunfish	7.5	9.0	6.0	4.0	26.5
(Lepomis megalotis)	Longear	6.0	10.5	8.0	1.0	25.5
(Lepomis gulosus)	Warmouth	9.0	9.0	5.5		23.5
(Micropterus salmoides)	Largemouth bass	6.5	12.0	2.0	2.0	22.5
(Ictalurus natalis)	Yellow bullhead	8.5	9.0	1.0	3.0	21.5
(Etheostoma chlorosomum)	Bluntnose darter	2.0	12.0	7.5		21.5
(Pomoxis nigromaculatus)	Black crappie	1.0	9.0	3.0	7.5	20.5
(Notropis emiliae)	Pugnose minnow		7.5	5.5	6.0	19.0
(Etheostoma asprigene)	Mud darter		9.0		9.0	18.0
(Elassoma zonatum)	Banded pygmy sunfish	12.0	2.0	4.0		18.0
(Notropis atherinoides)	Emerald shiner		7.5		10.0	17.5
(Etheostoma gracile)	Slough darter	9.0	2.0	5.0		16.0
(Lepisosteus oculatus)	Spotted gar	2.5	7.0	6.0		15.5
(Notemigonus crysoleucas)	Golden shiner	12.0	1.0	2.0		15.0
(Lepomis cyanellus)	Green sunfish	6.0	4.0	5.0		15.0
(Etheostoma proeliare)	Cypress darter	6.0	9.0			15.0
(Ictalurus punctatus)	Channel catfish		2.5		12.0	14.5
(Aplodinotus grunniens)	Freshwater drum		4.5	2.0	8.0	14.5
(Notropis fumeus)	Ribbon shiner	1.0	8.0	5.0		14.0
(Noturus gyrinus)	Tadpole madtom	6.0	6.0	1.0		13.0
(Pimephales vigilax)	Bullhead minnow				12.0	12.0
(Esox americanus)	Grass pickerel	9.0		2.0		11.0
(Amia calva)	Bowfin	10.0	S	1.0	S	11.0
(Fundulus notatus)	Blackstripe topminnow		9.0		1.0	10.0
(Notropis venustus)	Blacktail shiner		2.0		7.5	9.5
(Erimyzon sucetta)	Lake chubsucker	9.5				9.5
(Notropis texanus)	Weed shiner	6.0		3.0		9.0
(Dorosoma cepedianum)	Gizzard shad		3.0	4.0	2.0	9.0
(Centrarchus macropterus)	Flier	9.0				9.0
(Pylodictis olivaris)	Flathead catfish				8.0	8.0
(Ictiobus niger)	Black buffalo		2.0		6.0	8.0
(Hybognathus hayi)	Cypress minnow	5.5	1.0			6.5
*(Percina maculata)	Blackside darter				6.0	6.0
(Minytrema melanops)	Spotted sucker	1.5		2.0	2.0	5.5
(Micropterus punctulatus)	Spotted bass		1.0		3.5	4.5
(Cyprinus carpio)	Carp	S	1.5	S	3.0	4.5
(Lepomis symmetricus)	Bantam sunfish	1.5	1.0			2.5
(Hybognathus nuchalus)	Silvery minnow	1.5		1.0		2.5
(Ictiobus bubalus)	Smallmouth buffalo		2.0	S	S	2.0
(Pomoxis annularis)	White crappie	S	1.5			1.5
(Lepomis microlophus)	Redear	S	1.5			1.5
(Notropis maculatus)	Taillight shiner	1.0				1.0
(Labidesthes sicculus)	Brook silversides	1.0				1.0
(Ictalurus melas)	Black bullhead	1.0				1.0
(Lepisosteus platostomus)	Shortnose gar	S				0.0
(Ictiobus cyprinellus)	Bigmouth buffalo	S				0.0
(Esox niger)	Chain pickerel	S				0.0
	NUMBER OF SPECIES=	37	36	28	26	51.0

PROPOSED BIOLOGICAL CRITERIA FOR NEW YORK STATE STREAMS

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Abstract

Criteria are proposed for measuring significant biological alteration in New York State streams. The criteria are based on sampling benthic macroinvertebrate communities, and are site-specific, measuring change from conditions upstream of a given discharge. Sampling methods and the parameters on which the criteria are based are taken from EPA-proposed bioassessment methods which are presently being used in the New York State biological monitoring program. Replication in sampling is recommended to insure reliability of samples. Preliminary criteria were drawn from data sets collected from New York streams over a 5-year period. Sites designated as having significant biological alteration based on these criteria were cross-referenced with the Priority Water Problem List. Adjustments in the criteria were made to reflect levels consistent with the Priority Water Problem List. It is recommended that the criteria and methods be field implemented to insure proper levels of detectability.

Introduction

The Clean Water Act of 1972 was enacted with the objective "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters". To date, the objective of restoring the biological integrity of the nation's waters has been approached by regulatory agencies almost entirely through chemical and physical criteria. This manuscript advocates the adoption of biological criteria in the New York State regulations to monitor and restore the biological integrity of the State's waters.

The Water Quality Act of 1987 promoted the use of biological monitoring, stating that when numerical criteria are not available for 307 toxics, the "...state shall adopt criteria based on biological monitoring or assessment methods consistent with information published pursuant to Section 304(a)(8)".

In September, 1987, the USEPA Office of Water and Office of Policy, Planning, and Evaluation published the results of a major study of the Agency's surface water monitoring activities. The final report of this evaluation was titled, "Surface water monitoring: a framework for change". Of the six recommendation areas of this document, recommendations area 2 was to "accelerate the development and application of promising biological monitoring techniques". This recommendation was based on a conclusion that the chemical-specific approach cannot adequately protect all surface waters, and biological monitoring offers a tool to help manage surface water monitoring. In order to utilize biological monitoring in an important role in surface water monitoring, biological criteria must be proposed to determine if significant biological alteration is present, or if the biotic integrity of a given waterbody is achieving its full potential. This paper presents tentative biological criteria for New York State streams based on macroinvertebrates, and how they may best be applied to play a vital role in the State's surface water monitoring activities.

Table 1. Summary of presently known relationships between water quality categories and fish.

Category	Effect on Community
Non-impacted	Water quality not limiting to fish propagation or survival.
Slightly impacted	Water quality possibly limiting to fish propagation.
Moderately impacted	Water quality probably limiting to fish propagation.
Severely impacted	Water quality is limiting to fish propagation and survival.

New York State has long been a leader in the area of water quality monitoring and achieving water quality goals. At a recent EPA-sponsored National Workshop on the Development of Instream Biological Criteria, it was evident that one of the areas that will be in the forefront of water quality monitoring for the next few years will be the application of biological techniques to water quality standards. This is an important step in maintaining the States in a leading role in water monitoring.

Biological Criteria based on Macroinvertebrates

Freshwater organisms are frequently divided into 3 major groups: fish, periphyton (algae, bacteria, and rooted aquatic plants) and invertebrates. These 3 groups are each essential components of the freshwater ecosystem, and are interrelated in the aquatic food web. Of the invertebrates (animals lacking backbones), the majority are referred to as benthic macroinvertebrates, those that are bottom-dwellers, are large enough to be visible without a microscope, and are retained by a U.S. no. 30 sieve. The most common benthic macroinvertebrates are aquatic insects, worms, mollusks, and crustaceans.

Macroinvertebrates are capable of providing water quality information that chemical sampling cannot provide. Resident instream biota reflect the integrated effects of substances intermittently discharged, substances reacting synergistically with each other, or substances present in levels too low for chemical detection. They have potential in this regard for detection and evaluation for non-point problems.

Compared to other monitoring techniques, macroinvertebrate assessment is relatively rapid and inexpensive. With Rapid Assessment methods, a typical 5-site stream survey can be completed by a two person team in five days, including sampling, sample analysis, data analysis, and report writing.

The public relates better to biological criteria than to chemical criteria as measures of water quality. Especially organisms such as mayflies, stoneflies, and caddisflies are readily comprehended as clean-water indicators, fish-food organisms, and vital components of the aquatic ecosystem.

With the establishment of biological criteria, the detection of significant biological alteration could result in regulatory action. Benefits in this area would be substantial in instances of exclusive detection, in which chemical specific sampling was unable to detect a violation.

Biological Criteria and Use Impairment

The ultimate goal of the water quality program in New York State is to achieve and maintain the best usage for the State's waters. Of the best uses (e.g. drinking, swimming, fishing), biological criteria based on macroinvertebrates are most related to fish survival and propagation. Macroinvertebrates are connected to fish both as fish-food organisms and as indicators of damage to fish survival and propagation.

A recent study correlating fish and macroinvertebrates communities in New York State streams (Bode, 1987) found that levels of water quality impact based on macroinvertebrate communities are related to fish propagation and survival (Table 1). Since the proposed biocriteria are based on these levels of impact, they ultimately are also related to best use impairment. Although this does not imply that a significant biological alteration always indicates use impairment, it does provide basis for validity of macroinvertebrate standards as indicators of usage-related water quality problems.

Development of Biological Criteria

Sampling methods for wadeable streams with riffles followed "Methods for rapid bioassessment of streams (Bode 1988). These methods are nearly identical to protocol 3 methods as described in the proposed EPA Rapid Bioassessment Protocols (RBP). Using these methods 214 data sets from 27 streams in New York State were collected over a five year period from 1983 to 1987. Non-wadeable streams and streams lacking riffles were sampled using artificial substrate multiple-plate samplers, as described in Simpson (1980). Criteria based on artificial substrate samplers have not yet been developed.

Macroinvertebrate communities from the 214 data sets were assessed as to level of water quality impairment using a four tiered impairment system, as in EPA proposed RBP level 3. Based on five indicies (species richness, EPT, biotic index, dominant species, and field assessment) water quality was assessed as either nonimpacted, slightly impacted, moderately impacted, or severely impacted.

Changes between levels of impairment were averaged for all records and used to calculate significant biological alteration (SBA), the amount of change which would, on average, result in an increase in one level of impairment in a four tiered system (Fig. 1).

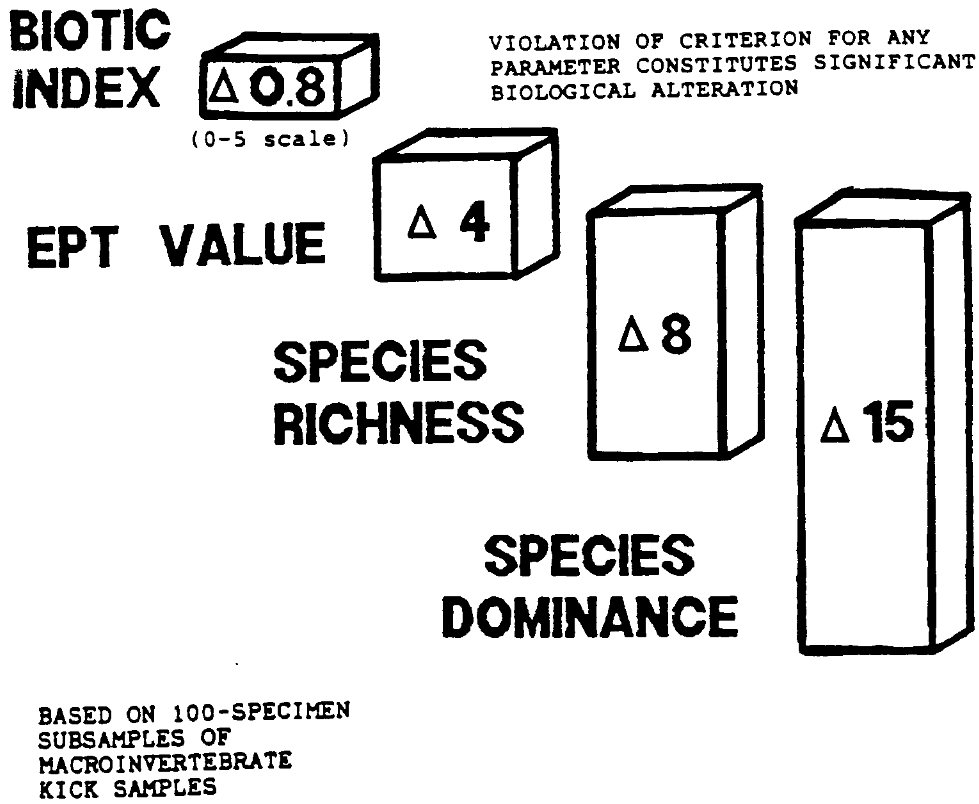

Fig. 1 Proposed criteria for assessing significant alteration of biota.

Stream sites which were indicated as having an SBA were cross-referenced with the Priority Water Problem (PWP) list. Of those not on the list, twelve were considered valid SBA's and new detection records, three were considered spurious, and six were not on the PWP list for valid reasons (out of state sites, and sites that were on the list at the time of sampling but have since been revoked). In practice, spurious detections could be invalidated by conflicting field assessments.

Although the rapid assessment methods utilized in 1983 to 1987 surveys are satisfactory for problem detection, problem assessment, and trend monitoring, the methods proposed for compliance monitoring differ somewhat, and have not yet been field tested. The primary difference in the two methods is sample replication, which should be required for compliance monitoring. With this method, three kick samples of two minutes duration would be taken, compared to the trend monitoring method of one kick sample of five minutes duration. Additionally, if compliance monitoring should become the responsibility of the discharger, greater variability among sampling methods will be introduced. Testing is necessary to determine if, 1). sample variability can be controlled for three replicate kicks, and 2). the proposed criteria will be supported by three replicate samples as they were for single (non-replicated) kick sampling. It is strongly recommended that the proposed criteria be tested over a range of water qualities in several stream systems using the proposed sampling procedure.

Specifications of Proposed Biological Criteria

Sampling methods -- Sampling methods for wadeable streams with riffles will follow those described in "Methods for Rapid Biological Assessment of Streams (Bode, 1988). These methods conform to the EPA-proposed Rapid Bioassessment Protocol three. To insure sample validity, this protocol will be modified for compliance monitoring by collecting three replicate kick samples of two minutes duration at each site. Sampling methods for non-wadeable streams and streams lacking riffles will follow artificial substrate methods as described in the EPA biological field and laboratory methods (USEPA 1973), and as modified for use in New York State streams as described in Simpson (1980), and are currently under development.

Site selection -- The proposed criteria are site-specific. Using the paired site comparison method (Green, 1979), significant biological alteration will be measured from a control, either upstream, or if no suitable upstream site is available, from a comparable nearby stream. Sites are selected to have similar current speeds, substrate particle size, degree of embeddedness, and percent canopy.

Significant biological alteration — Significant biological alteration is measured as statistically significant change from a control site which exceeds allowable levels of change in any of 4 parameters (Fig. 1). These parameters, species richness, biotic index, EPT value, and change in species dominance, were selected from the 8 parameters listed under the EPA RBP Protocol level three. The parameters were selected on the basis of: 1). ability to accurately detect a wide range of biological alterations, and 2). ability of the parameter to be simply computed and comprehended. Levels were determined for each parameter based on the amount of change measured between the 4 levels of impact in a four tier system.

The parameters and the proposed criteria for each are as follows:

Species Richness: The total number of species or taxa found in the 100-organism subsample. The criterion for this parameter is eight; a decrease of eight or more species constitutes significant biological alteration.

EPT Value: This index from Lenat (1988) is the total number of species of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichopetera) found in the 100-organism subsample. These are considered to be mostly clean-water organisms, and their presence generally is correlated with good water quality. The criterion for this parameter is four; a decrease of four or more constitutes significant biological alteration.

Biotic Index: The Hilsenhoff Biotic Index is calculated by multiplying the number of individuals of each species by its assigned tolerance value, summing these products, and dividing by the total number of individuals.

Tolerance values are listed in Hilsenhoff (1982, 1987). On a zero to five scale, tolerance values range from intolerant (0) to tolerant (5). The criterion for this parameter is 0.80; an increase of 0.80 or more constitutes significant biological alteration.

Dominant Taxon Contribution, or Species Dominance: This number is the percent contribution of individuals of the most numerous taxon or species in the sample. High percent contributions are an indication of community imbalance. The criterion for this parameter is 15; an increase of 15 or more percent constitutes significant biological alteration.

Application of the Proposed Biological Criteria

In application, the following tentative schedule would be the recommended procedure for implementing biocriteria. Sampling is conducted at a comparable control (upstream) site and an impact (downstream) site by a trained biologist. Sampling could initially be conducted by NYS DEC personnel, or ultimately be the responsibility of the discharger. Three replicates are collected at each site, with equal effort being assured at control and impact site.

All three replicates from each site are laboratory processed and identified, using 100 specimen subsamples from each. The four parameters (species richness, biotic index, EPT, and species dominance) are calculated for each of the replicates.

To determine if the replicates are within the proper range of variability, similarity coefficients are calculated between replicates. If variability exceeds the maximum amount, additional sub-samples are taken and processed; if variability between replicates persists, sites are resampled.

The values for each parameter are screened to determine if any exceed the criterion for that parameter. An experienced biologist examines these values to check for spurious results.

Valid sites having significant biological alteration are reported to NYS DEC.

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RAPID BIOASSESSMENT USING FUNCTIONAL ANALYSIS OF RUNNING WATER INVERTEBRATES

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Introduction

Freshwater bioassessment approaches using benthic macroinvertebrates typically have focused on the questions of "what is it?" and "how many are there?" Most frequently, the answers to these questions are phrased in the form of numerical indices. Clearly the answer to the first question is dependent upon how well the identity of the species collected can be distinguished. Although most biosurvey reports, as well as most of the published literature, purport to present evaluations of "species" diversity, richness, or some other index, most frequently only a small faction of the specimens are ever actually classified to species. Almost without exception, the indices are based on a list of miscellaneous taxa ranging from class or order through families and genera, and occasionally to the species level. Often, as in the case of the Chironomidae, there is an inverse relationship between numerical abundance and taxonomic detail. Even though it may be intellectually satisfying to view the species as the basic unit in freshwater benthic ecology, it is at present operationally impossible. Because of present limitations, it seems prudent to consider alternate strategies. First, the focus of assays can be changed from "what it is" to "what it does." Second, the question of "how many", which requires at least semi-quantitative samples, such as kick samples or catch per unit effort methods, can be changed to "relatively how many" which permits the use of qualitative samples expressed as dimensionless ratios. The relative abundance of invertebrates categorized by their functional roles in aquatic ecosystems is the basis of the approach discussed below.

Functional Organization of Benthic Macroinvertebrate Assemblages

It should be noted that the functional approach has been incorporated in various forms into a large number of research projects published in the open literature (e.g. Cummins et al. 1982; Minshall et al. 1983). In these studies, professional freshwater researchers (including advanced degree students and technicians) made the value judgments necessary to complete the analyses. I contend that if the necessary expertise is available to use numerical indices based on various levels of taxonomic determination, it must be available for the functional analysis of benthic communities.

The basic tenets of the functional approach are that: 1) the focus is on the functional roles played by macroinvertebrates in freshwater ecosystems, and 2) taxonomic effort is directed toward maximum resolution of these functional roles. A major consequence of the approach is that no effort is expended in making taxonomic separations below the level that allows a functional designation. For example, determination at the ordinal level of Odonata and Megaloptera is sufficient to designate the category of functional feeding group, namely predator, at nearly the 100% confidence level (the exceptions being newly hatched first instars (e.g. Petersen 1974).

Table 1. Functional modules relating food resource categories and morpho-behavioral food acquisition groups of freshwater macroinvertebrates

MODULE	DESCRIPTION
Shredders-CPOM	Coarse particulate organic matter (CPOM, particularly leaflitter) - Microbes (especially aquatic hyphomycete fungi) -Shredders (invertebrates with chewing mouth parts)
Collectors-FPOM	Fine particulate organic matter (FPOM, including shredder feces) - Microbes (dominated by bacteria) Collectors: Filtering Collectors (invertebrates that remove particles from suspended load); and Gathering Collectors (invertebrates that gather particles from sediment surfaces or interstices or associated structure)
Scrapers-Periphyton	Periphyton (attached algae, especially diatoms, and associated micro flora and fauna) - Scrapers (invertebrates with adaptations for removing algae from relative hard planar surfaces)
Predators-Prey	Prey (invertebrates available for capture) - Predators (invertebrates specialized to capture and consume, at least partially, live prey)

Similarly, designation of the ephemeropteran family Heptageniidae establishes the scraper functional feeding group at about a 95% confidence, exceptions being species that function as filtering collectors or shredders on wood or leaf litter (Merritt and Cummins 1984). Although it is generally true that in some groups, such as the Chironomidae, the more refined the taxonomy the better the resolution in assigning animals to functional groups, it is important to consider how much increased resolution accrues from additional effort spent in taxonomic determinations. For example, unlike the heptageniid mayflies, separation of ephemerellids into genera (recently elevated from subgeneric status) does confer significant increases in functional group resolution (Edmunds et al. 1976; Merritt and Cummins 1984). Of course samples can be preserved and taxonomic analyses performed at a later date. However, the method is designed primarily for use in the field (Cummins and Wilzbach 1985) so that evaluations can be completed, or the data gathered that will allow the evaluations to be made, before leaving the stream or lake site.

It is important to emphasize that the functional group designation should be made as the initial determination using an approach such as that given by Cummins and Wilzbach (1985) with more detailed taxonomy as a second step. Presently the most common approach is to make taxonomic determinations on preserved specimens, which of course excludes the use of many

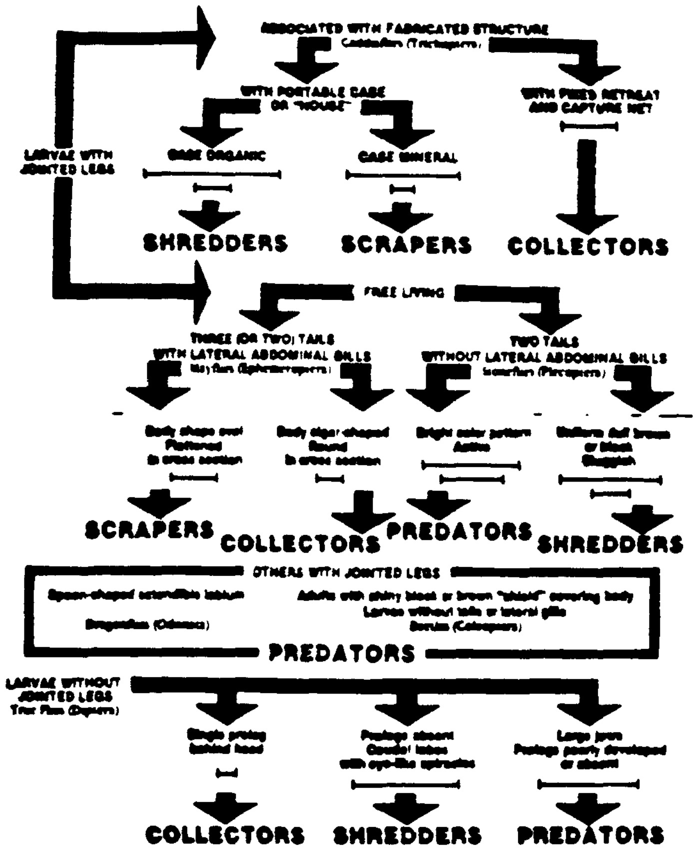

Fig. 1. Summary of the separation of freshwater macroinvertebrates into functional feeding groups from Cummins and Wilzbach 1985. The bars shown indicate the size ranges of full grown (terminal instars) animals representative of each category.

helpful behavioral and natural color pattern characters, and later assign functional groups using the ecological tables in Merritt and Cummins (1984). A major difficulty in the assignment of functional group designations has been the continued interchangeable use of two concepts that are not synonymous.

One concept is the designation of trophic levels, that is, herbivores, detritivores, and carnivores. These designations are based on digestive tract analyses. Such gut inventories convey little information about the mechanisms involved in acquiring the food. For example, the presence of large amounts of fine particulate detritus in the digestive tract can result from the material being filtered from the passing water, mined from the interstices of the sediments or brushed from surfaces, or harshly scraped from sediment surfaces coated with sparse attached algae and detritus trapped among the cells and colonies. Thus, in this example, the food type, fine detritus, is similar but the morphological-behavioral mechanisms by which it was acquired would be fundamentally different.

The other concept, functional feeding groups, is based on the system responsible for the <u>acquisition</u> of the food. Acquisition systems are separated on the basis of morphological structures and associated driving behavior according to general classes of food resources that are most efficiently harvested by a given functional group (e.g., Cummins and Klug 1979). A corollary of this concept is that the efficiency with which the acquired food is converted to growth is dependent on the coupled assimilation system (Cummins 1984).

A clear example of the difference between the two concepts can be found in Cummins (1973) where data on the scraper caddisfly Glossosoma nigrior is presented. Populations of G. nigrior from two streams had distinctly different gut contents. Larvae collected from an algal rich stream had gut contents composed of 90 to 99% algae, while those from an algal poor stream had digestive tracts containing 48 to 93% detritus. Larvae from the two streams employed the same acquisition system. They fed by scraping rock (cobble and gravel) surfaces, but they would have been classified as herbivores in one stream and detritivores in the other. Similar studies with glossosomatid caddisflies in Michigan and Oregon confirmed this type of variation in gut contents in different populations all of which were feeding as scrapers (Anderson and Cummins 1979).

Based on gut contents (the trophic system), essentially all aquatic macroinvertebrates are omnivores if all growth stages under a variety of habitat and seasonal conditions are considered. Virtually all early stages (instars) of all species ingest fine particulate organic matter (FPOM) detritus. FPOM includes such items as fine macrophyte fragments, amorphous organic matter (a significant portion being derived by co-precipitation of dissolved organic matter), bacteria, fungal hyphal fragments and spores, protozoans, rotifers, nematodes, diatoms, green and bluegreen algae, microcrustaceans and a variety of other microinvertebrates, and first instars of some macroinvertebrates. The most apparent continuity, given such a diverse food resource, is the nature of the morpho-behavioral mechanisms employed in acquiring such a slurry food source from the water column or the sediments. If a particle of this "detrital porridge" is above a certain size (probably about 1 mm) it will need to be chewed up perhaps in addition to, but usually instead of, the filtering and gathering mechanisms alluded to above.

The basic functional modules linking food resource categories and morpho-behavioral food acquisition groups are summarized in Table 1.

As the relative dominance of the various food resource categories changes, there is a corresponding shift in the ratios of the different functional groups. For example, such adjustments between food resources and macroinvertebrate functional groups was a basic component of the River Continuum Concept (Vannote et al. 1980; Minshall et al. 1983, 1985; Cummins et al. 1984). The adjustments were shown to vary with increasing stream and river size (order) from headwater tributaries to larger rivers (Cummins et al. 1981).

Functional Group Analysis

The method of analysis recommended is that presented in the field manual by Cummins and Wilzbach (1985). This procedure is intended for use in the field with freshly collected live samples. Following this rapid, field bioassessment, samples can be preserved for more detailed taxonomic analysis in the laboratory. A general summary of the functional group separations from Cummins and Wilzbach (1985) is given in Figure 1.

The procedure is based on qualitative samples taken from representative materials in five general categories: leaf and other plant litter, large cobble, fine sediments from depositional areas, large wood and rooted plants. Because the results are calculated as dimensionless ratios, these habitat-food resource samples need not be quantitative. The procedure of functional group analysis given in Cummins and Wilzbach (1985) is organized by two levels of resolution. The first level provides for functional group separations at a level of efficiency in the range of 75 to 100%, depending on the groups involved. The second level of resolution, which requires more taxonomic expertise than the first level, increases the resolution another 5 to 10%, again depending upon the group in question, by dealing with some of the most commonly encountered exceptions. In particular watersheds in which repeated monitoring would be conducted, additional levels of resolution could readily be devised.

In evaluating the relative abundance of functional groups as dimensionless ratios in a given aquatic ecosystem, several cautions are important. First and foremost, it is important to establish some ratios for one or more reference systems in a given drainage basin. The data for these ratios should come from the most undisturbed streams in the basin and take into account stream size (order). As shown in the River Continuum studies, the ratios change in a predictable fashion with increasing stream order along a drainage basin network (Cummins et al, 1981; Minshall et al. 1983).

A second caution involves the seasonal timing of sampling. For functional group analyses, and any taxonomically based numerical index as well, it is important to sample a given stream or river when the greatest number of species are present and feeding, and of as large a size as possible. In the Temperate Zone this usually means avoiding the late spring through early summer and the early autumn through mid-winter periods. During these periods many species are present as eggs or newly hatched immatures, or are entirely absent from the stream as terrestrial adults. It should be noted also that there are definite spring-summer and autumn-winter communities of invertebrates made up of populations with feeding periods concentrated in one or the other (Cummins et al. 1989). This means that at least two sampling times each year would be advisable.

The above cautions are discussed in Merritt and Cummins (1984) and Cummins and Wilzbach (1985), and some examples of expected ranges in ratios by stream order are given in the latter. However, as stated above, the expected ratios need to be regionalized to avoid the "universally applicable index" syndrome. Clearly, functional feeding group analysis is not a substitute for detailed taxonomic studies of stream macroinvertebrates, but it has been used successfully to gain insight into the organization of freshwater communities as they are influenced by changes in the resource base.

In summary, I offer the following epilogue.

An Ode to Rapid Bioassessment

To inquire on the health
  Of our native stream wealth,
We seek rapid assessment

To reduce our investment.
Never mind if we blunder
Just give us a number.
Treat taxonomy chaotic
With an index biotic.
Is the system redundant?
Just rank midges abundant.
Is it a group you despise?
Use a coarser mesh size.
Never mind all the species
That's a whole different thesis.
So...with a lumper's devotion
I offer this motion-
Take a moment or two
Try the functional view.

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A STREAM INVENTORY PROCESS TO CLASSIFY USE SUPPORT AND DEVELOP BIOLOGICAL STANDARDS IN NEBRASKA

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Abstract

The Nebraska Department of Environmental Control (NDEC) is currently conducting a five year inventory and classification process of the state's perennial streams. The foundation of this effort is the collection of physical and biological data by major river basin. This process consists of four phase approach including a review of existing data, reconnaissance survey, biological collection and standards revision. Once all perennial streams are identified in a basin using topographic maps (1:24,000), a reconnaissance survey is conducted. This includes a field visit to each perennial stream. Physical stream factors such as flow, channel characteristics, width, depth, substrate, streambank stability, habitat quality; and surrounding watershed factors such as land use, buffer zone characteristics and soil type are documented. Upon completion of the the reconnaissance survey, sites are selected to collect biological data. These biological collections include both qualitative fish and macroinvertebrate collections representing the various stream types found in a basin. Emphasis is placed on selecting "least impacted" reference streams so a benchmark can be established for interpreting biotic integrity. The final phase includes use class assignments and standards revision for the redefined stream segments. This substantially increases the number of stream segments but provides better water quality protection. Key fish species and cold water indicators are currently being used to assign aquatic life use classes to each segment. The biological data collected will be used to identify faunal regions and develop biotic assessment tools. This process has other applications including nonpoint source assessment, construction grants prioritization and identification of impaired instream beneficial uses.

Introduction

The Clean Water Act (CWA) of 1977 was ammended in 1981 to authorize grants to states for water quality management planning under section 205(j). The major effort of this program in Nebraska is the inventory and classification of all perennial streams (totaling some 10,212 miles) based on their physical and biological characteristics. This approach is targeted at documenting the beneficial uses of a stream which is defined as "an existing use of a water body or one that is attainable based on the physical, chemical or biological water body characteristrics (NDEC 1987). Beneficial uses recognized by NDEC include recreation, aquatic life, water supply, aesthetics and public health (Table 1). Past monitoring by the NDEC had been primarily traditional ambient chemical sampling. This indirect assessment of the state's stream resources has fallen short in assessing whether waters are "fishable and swimmable" as mandated by Congress in the CWA. Others including Thurston et al. (1979) and Karr and Dudley (1981) have also concluded that aquatic resource assessments cannot be based solely on physiochemical sampling. According to the CWA one

Table 1. Beneficial uses of surface waters recognized in the Nebraska Water Quality Standards (NDEC 1987).

Type	Description
RECREATION	Based on physical stream characteristics[a]
Class A	Full body contact
Class B	Partial body contact
AQUATIC LIFE	Based on habitat, stream size and presence of fish species and other aquatic life
Coldwater	Seldom exceeds 25 C and coldwater biota present[b]
Class A	Self-sustaining salmonid fishery
Class B	Presence of two or more coldwater indicators including flora and fauna
Warmwater	Often exceeds 25 C and warmwater biota present
Class A	Flow greater than 10 cfs and habitat good, or one or more key species present year round[c]
Class B	Flow less than 10 cfs and habitat limited, no key species supported year round
WATER SUPPLY	
Public Drinking Water	Numeric
Agricultural	Livestock and irrigation
Class A	Numeric conductivity and nitrate
Class B	Naturally limited
Industrial	Narrative depending on specific use
GENERAL AETHETICS AND PUBLIC HEALTH	Narrative

[a] width, depth, substrate and access; [c] See Table 3.
[b] As identified by NDEC;

of the objectives is to maintain the chemical, physical and biological integrity of the nations waters. Unfortunately, of these three characteristics biological monitoring has received the least attention, but perhaps is the most important because organisms integrate all surrounding environmental effects. They can be thought of as a continuous barometer of water resource quality.

Prior classification methods of Nebraska streams have lacked any standard procedures or accountability system. This type of information is necessary when justifying the need for advanced treatment by permitees or classifying a stream as an outstanding resource water.

Classification of aquatic life is lacking in Nebraska with the exception of a few research projects and a fishery survey conducted by Nebraska Game and Parks Commission (NGPC) (Bliss and Schainost 1973). A stream classification map was developed by NGPC (1978) to assist state and federal agencies and water users in assessing the impacts of proposed water projects. The map displays an appraisal of the relative value of stream fishery resources. Most of the stream fishery data collected has been for sport fish appraisal and management with little work on habitat quality or community assessment. Current aquatic life data is needed on what is attainable in the various types of

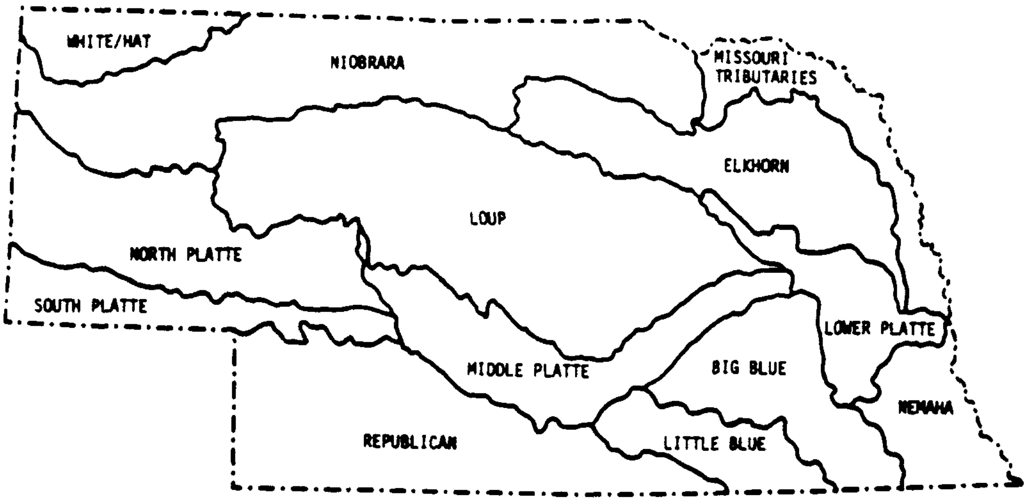

Fig. 1. Nebraska's thirteen major river basins.

streams in Nebraska. This community data could then be used as a benchmark to assess "biotic integrity" defined by Karr and Dudley (1981) as the "ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region."

The objectives of this five year effort are to develop: 1). a systematic, scientific approach to classify perennial stream resources according to existing or attainable uses; 2). biological assessment techniques to directly measure aquatic life health based on regionally expected fauna, and 3). collected current data directly applicable to standards revisions, construction grants prioritization, nonpoint programs and reporting of impaired waters required by the 305(b) biennial water quality report.

Methods and Materials

The inventory and classification process is a four phase approach including a review of existing data, reconnaissance survey, biological collections and standards revision. The 13 major river basins in the state (Fig. 1) provide a logical framework to conduct a systematic sampling over a scheduled five year period (1984 to 1989). This process began in the Elkhorn Basin during 1984, which served as the prototype for the remainder of the state. This basin was selected first because it offered the greatest variety of stream types, representing four ecoregions according to Omernik's (1986) classification map. Due to internal budgetary constraints, three basins will be completed each

Table 2. Habitat variables measured during field reconnaissance surveys.

Habitat Quality	Riparian Zone
Temperature (coldwater or warmwater)[a]	buffer quality
Cover occurrence	% vegetation (trees, shrubs grass)
Pool quality	
Riffle/run occurrence	land use
flow class[b]	channelized (yes or no)
substrate type[c]	bank stability
average depth[c]	sand bar occurrence[d]
average width[c]	streambank (sand or dirt)[d]

[a] As determined by water temperature and aquatic life indicators including aquatic flora and fauna.

[b]

Flow class	cfs	Flow class	cfs
1	0.1 < x <1	6	51-100
2	1-5	7	101-250
3	6-10	8	251-500
4	11-25	9	> 500
5	26-50		

[c] Estimated by cross section transects

[d] Used to document potential for partial and full body recreation uses along with flow, stream width, depth and substrate.

of the remaining four years until the entire state is classified.

The following steps summarize each phase of the inventory and classification process.

Phase 1 -- Review Existing Data -- All perennial streams of a basin are identified from topographical maps (1:24,000). Sites are then selected to represent homogeneous segments which are identified using flow records (i.e., Department of Water Resources and United States Geological Survey), physical conditions (e.g., substrate and land use), fishery data gathered by NGPC and proximity of a stream to other tributaries which may change the flow regime.

Phase 2 -- Reconnaissance Survey -- Once stations are selected, each site is visited by a two man team, with at least one being a trained biologist. Qualitative descriptions of habitat are made by assessing specific variables both by observation and transect measurements. Approximately one-

Table 3. Key fish species used in the classification of aquatic life uses.

Common Name	Scientific Name
Threatened	
Lake Sturgeon	_Acipenser fulvescens_
Pallid sturgeon	_Scaphirhynchus albus_
Northern redbelly dace	_Phoxinus eos_
Finescale dace	_P. neogaeus_
Pearl dace	_Semotilus margarita_
Blacknose shiner	_Notropis heterolepis_
Sensitive	
Lake chub	_Couesius plumbeus_
Brook stickleback	_Culea inconstans_
Iowa darter	_Etheostoma exile_
Johnny darter	_E. spectabile_
Orangethroat darter	_E. spectabile_
Blacknose dace	_Rhinichthys atratulus_
Grass pickerel	_Esox americanus_
Recreationally Important	
Shovelnose sturgeon	_Scaphirhynchus platorynchus_
Paddlefish	_Polyodon spathula_
Brook trout	_Salvelinus fontinalis_
Brown trout	_Oncorhynchus trutta_
Rainbow trout	_O. mykiss_
Northern pike	_Esox lucius_
Muskellunge	_E. masquinongy_
Blue catfish	_Ictalurus furcatus_
Channel catfish	_I. punctatus_
Yellow bullhead	_Ameiurus natalis_
Flathead catfish	_Pylodictis olivaris_
Striped bass	_Morone saxatilis_
White bass	_M. chrysops_
Rock bass	_Ambloplites rupestris_
Largemouth bass	_Micropterus salmoides_
Smallmouth bass	_M. dolomieui_
Spotted bass	_M. punctulatus_
Bluegill	_Lepomis macrochirus_
Redear sunfish	_L. microlophus_
Black crappie	_Pomoxis nigromaculatus_
White crappie	_P. annularis_
Yellow perch	_Perca flavescens_
Sauger	_Stizostedion canadense_
Walleye	_S. vitreum_

half hour is spent on each site visit. Direct measurements of physical characteristics and variables are assessed (Table 2). Definitions used for the variables were compiled from Armour et al. (1983) and Platts et al. (1983). Three transects are usually made of the study reach which is 20 to 40 times the stream width. Stream width is measured at each of three points at each transect to the nearest 0.5 ft. Stream depth is measured to the nearest 0.1 ft at a series of equally spaced intervals across the stream. Substrate is visually estimated to the nearest 10% using the categories recognized by Platts et al. (1983). Photographs upstream and downstream of the study reach (usually a bridge crossing) are taken for further reference. To achieve consistency all variable measurements are recorded on a standard field form.

<u>Phase 3 -- Biological Survey (Fish and Macroinvertebrate)</u> -- Biological sites are selected by the staff who performed the field reconnaissance on a particular basin. The selection process is targeted at least impacted reaches which will serve as reference streams (e.g., wildlife management areas, refuge or state park). Collections from these stations will document the best attainable or "best expected" community found in a basin. The selected sites include the various stream types found in a particular basin. Flow class, substrate, temperature class (warmwater or coldwater) and habitat quality are the primary characteristics considered to represent the different stream types. The selection process also considers critical reaches defined as having ecologically significant

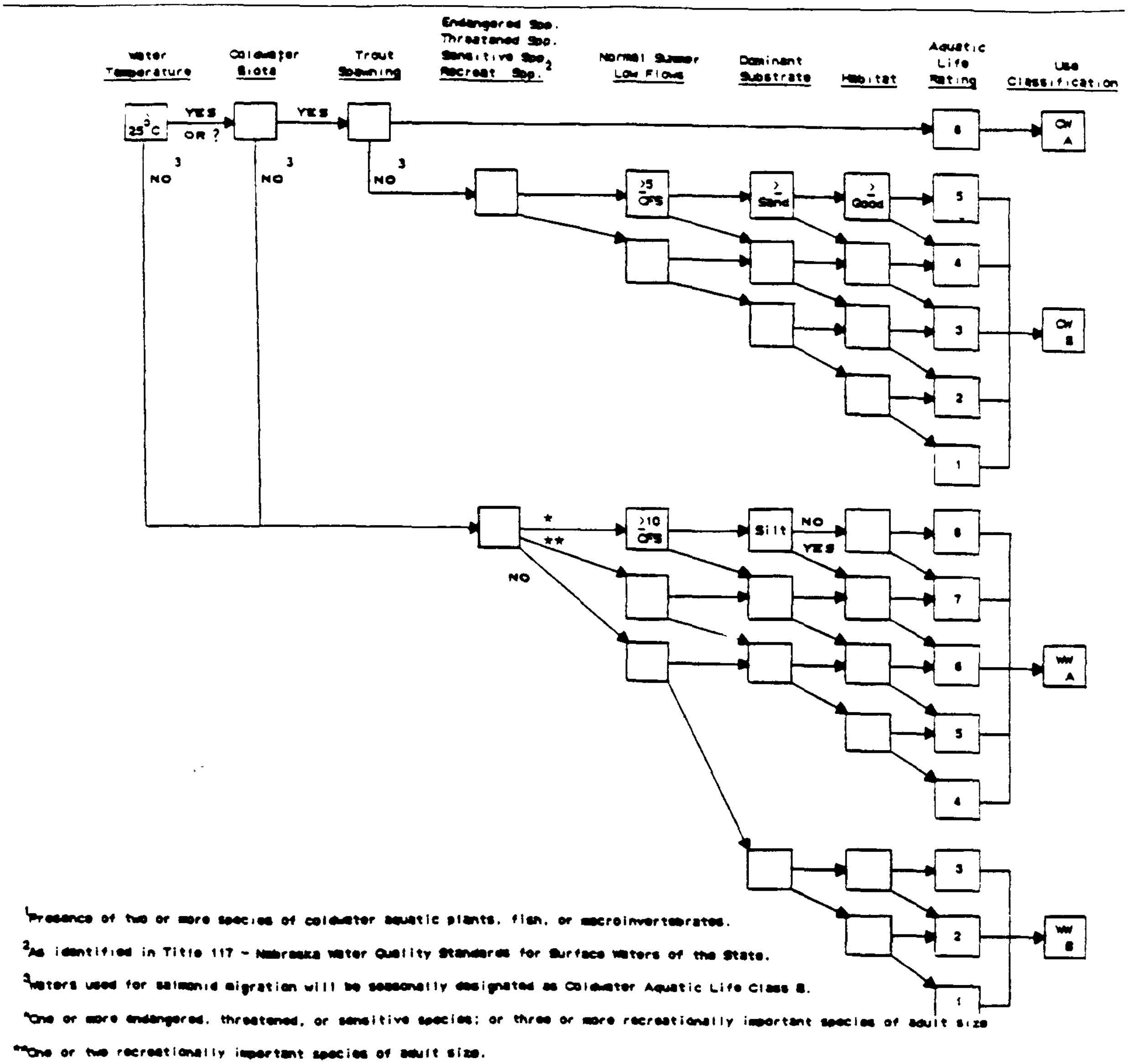

Fig. 2. Decision tree used by the Nebraska Department of Environmental Control to classify aquatic life uses of perennial streams in Nebraska.

fish species (e.g., salmonid spawning, threatened or sensitive categories) documented within the last 20 years. Stream reaches with these suspected attributes are sampled to document the occurrence of these key fish species (Table 3).

Once sites are selected a field crew of five people collect qualitative samples of both macroinvertebrates and fish. Samples are gathered during summer months (June 1 to September 15) when flows are stable and fish migration movements are at a minimum. High flows or unusual conditions potentially affecting the community or sampling effectiveness are avoided.

A crew of three people using electrofishing gear sample all available habitats at a distance at least 40 times the width. Sampling is restricted to wadeable streams;

Table 4. Example of the Stream Habitat Quality Index Used to Evaluate Nebraska Streams.

RATING ITEM	CATEGORY (Circle the appropriate score for each)			
Instream Cover	Common (6)	Occasional (4)	Rare (2)	None (0)
Bank Stability	Excellent < 10% eroded banks (6)	Good 11-30% eroded banks (4)	Fair 31-50% eroded banks (2)	Poor > 50% eroded banks (0)
Flow Fluctuation	Minor little or none from base flow (4)	Moderate Evidence of debris along middle portion of banks (2)	Severe Evidence of debris high on banks (0)	Severe Intermittent Stream (0)
Riffle/Runs	Common (3)	Occasional (2)	Rare (1)	None (0)
Pool Quality	Class 1 or Class 2 Pools Common (3)	Class 2 Pools Occasional or Rare (2)	Class 3 Pools Common or Occasional (1)	Class 3 Pools Rare or Class 4 Pools only (0)
Bottom Substrate	> 50% gravel or larger substrate (3)	31-50% gravel or larger (2)	10-30% gravel or larger substrate (1)	Bottom uniform shifting sand hardpan clay bedrock or silt bottom (0)

Total Score For All Categories	
20 - 25	Excellent
15 - 19	Good
10 - 14	Fair
< 10	Poor

extremely large rivers (e.g., Missouri River) are not sampled. Fish species are identified and enumerated in the field in most cases. Total length is recorded on all recreationally important species to document possible nursery streams having natural reproduction. Representative specimens of most species are preserved for voucher specimens for the University of Nebraska museum and further identification made in the laboratory if necessary. Field notes including sampling time to estimate abundance, anomalies of individuals (e.g. disease and parasites) and habitat conditions are made. In situ measurements of dissolved oxygen, conductivity, pH, temperature and flow (measured by a Pygmy meter) are also made at each biological site to characterize the physiochemical environment.

A crew of two people sample the macroinvertebrate community at all available habitats using various collection methods (e.g. kick sampling, debris picking and sieving sediment or detritus). At least 100 individuals of a particular taxon are gathered before the field sampling is shifted to other less common taxa at the site. Usually, one man-hour is spent in the field to sample all habitats. Samples are concentrated in a sieve, preserved with 10% formalin and returned to the lab for identification. Samples are sorted and specimens identified to the lowest practical level (usually genus). Large samples and chironomids are often subsampled. Normally at least a quarter of the sample is used to extrapolate total numbers. Common taxa are identified and counted up to 100 individuals. Field notes are made of habitats sampled, substrate types and habitat quality in addition to any peculiarities observed at the site.

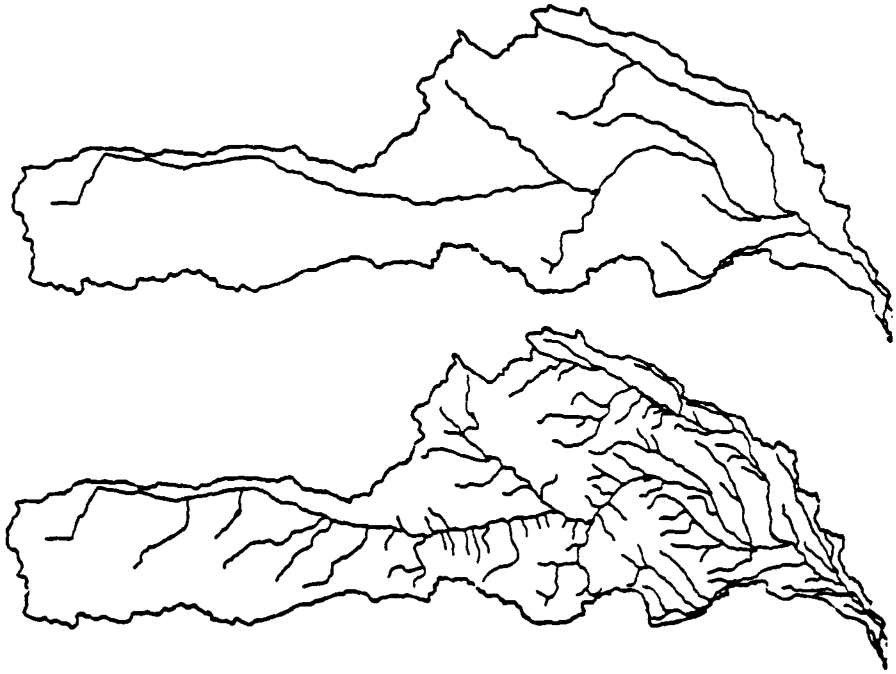

Fig. 3. The Elkhorn basin showing the stream segments delineated before (above) and after (below) the inventory process.

Phase 4 -- Standards Revision -- Once all data has been collected for a particular basin, stream segments are identified. These segments are catalogued by four hierarchial levels including river basin, subbasin, stream and stream segment. Existing and attainable beneficial uses for each stream segment are determined (Table 1). A decision tree is used in making a determination of use classification for aquatic life (Fig. 2). Similar decision trees have been constructed for state resource waters, recreation, water supply, agricultural and industrial uses.

All data is compiled, cataloged and each stream segment categorized in a stream inventory document. This document serves as a reference and accountability system for all streams sampled.

Results and Discussion

In the Elkhorn basin a total of 1,770 mi of stream were represented by 250 reconnaissance sites. These streams were later classified into 134 homogeneous stream segments based on physical and biological characteristics, compared with 18 before the process (Fig. 3). This substantially increases the number of segments but provides better protection to the resource because segments are described by individual characteristics. In addition, 27 reference sites were sampled to establish expected biological communities for the various stream types in the basin.

Upon completion of the statewide biological collections this data will be utilized to identify aquatic faunal regions using methods outlined by Whittier et al. (1987) and Hughes and Larsen (1988). Aquatic species typically found in the various regions, as indicated by frequency of occurrence, in addition to species indicative of specific stream types will be listed. This regionalization will greatly reduce the natural variability observed in biological communities over a large geographic area having a wide variety of stream types. This will facilitate the development of biotic assessment tools specific to a particular type of stream and region.

BIOS, a national Storet system maintained by USEPA is currently being used to store biological data along with most habitat and physical information. This data can be integrated with SAS to automate data assessment and also serve as a national data bank for other users.

A list of fish and macroinvertebrate species will be compiled listing ecological characteristics (e.g. tolerances, trophic class and coldwater adapted). This list will be compiled by NDEC using the acquired survey data, regional ichthyofaunal references and expert review from outside sources.

Once aquatic faunal regions have been identified and expected faunal lists finalized, then biotic indicies can be developed. The index of biotic integrity (IBI) devleoped by Karr et al. (1986) using fish communities provides a sound approach at assessing biotic integrity using fish. This index provides valuable metrics encompassing species and trophic composition, abundance, and condition, all of which are ecologically related to stream health. Its value in assessing biotic integrity as it relates to water resource degradation has been demonstrated in the midwest (Angermeier and Schlosser 1987). Some modifications of the IBI tailored to the identified regions with similar fish faunas have been developed (Miller et al. 1988). Similar metrics which best indicate degradation of the macroinvertebrate community integrity will also be formulated into indicies.

Eventually, numeric biological criteria can be developed once enough data has been collected on reference streams to describe the variability expected in the various regions. Percent deviation from expected scores for the various aquatic life uses can initially be developed as indications of community health and use support. Until such time, the standards will reflect primarily key fish species occurrences and indicator species (i.e. coldwater fauna).

Surrogates to stream size such as stream order or drainage area have been used to account for the effects of stream size on the number of species found (Karr et al. 1986; Leonard and Orth 1986). NDEC is currently investigating flow class (Table 2) as a surrogate to stream size to establish "maximum species richness" lines. Stream order and drainage area appear to be of limited utility in some Nebraska basins such as the Sandhills where groundwater contributions can be excessive.

Some measures of habitat quality which summarizes the observations and measurements taken in the field is necessary when collecting biological data. Habitat is extremely important in determining the attainable aquatic life uses of a stream. If a stream has sufficient flow and habitat quality but does not support the expected aquatic life, it can be inferred that water quality may be limiting. Habitat assessment techniques outlined by Ball (1982) and Platts et al. (1983) were used to develop this index. A habitat quality index designed for Nebraska waters is outlined in Table 4. Variables were weighed according to their importance in explaining the diversity and abundance of aquatic life found in Nebraska. A habitat rating for a particular stream reach used in conjunction with biological sampling can be valuable in determining the existence of a water quality problem.

Other important uses for this inventory and classification process in water quality management and planning activities include a nonpoint assessment report mandated by section 319 of the CWA of 1987. Reconnaissance data including riparian zone condition, streambank stability and field observations were utilized in this assessment report. Future prioritization of watersheds requiring nonpoint abatement programs will utilize fishery data (key species) and physical factors measured during reconnaissance surveys (i.e. swimming use existing or attainable) to arrive at a consistent ranking mechanism. Construction grants prioritization takes into account existing or attainable recreation or aquatic life uses. This is necessary to justify the expense of upgrades or the type of treatment required (e.g. chlorination). It can be used for determination of mixing zones which are used in conjunction with wasteload allocations. The physical data gathered from the various stream segments inventoried is utilized to assist in these cases. Certification of water quality as part of the 401/404 program. This utilizes both physical and biological data gathered to make determinations on whether an applicant is denied approval on a particular project. It is useful for tracking trends in ambient surface waters. Reference streams as identified by this inventory as the "best expected" aquatic life for a region can serve as the focal point for long term trends if monitored routinely. It is useful for satisfying the requirement of the antidegradation clause of Nebraska's Surface Water Quality Standards. Recent biological data can be used as evidence to identify outstanding or unique resources requiring more stringent protection. Finally, it can be useful for identification of surface waters not fully supporting their designated uses, a requirement of the 305(b) report.

The 205(j) stream inventory process is an integral part of most surface water quality management programs. It provides a strong basis for making decisions relating to Nebraska stream resources. This process will not only identify problem areas where treatment is needed but also provide framework to develop biological assessment tools. Ultimately, this process will provide better protection of Nebraska's stream resources by furnishing information and evaluation tools targeted at instream beneficial uses.

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THE USE OF BIOSURVEY DATA IN THE REGULATION OF PERMITTED NONPOINT DISCHARGERS IN VERMONT

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Abstract

High quality upland streams in Vermont are being increasingly threatened by the use of nonpoint effluent spray and leachfields around developing resort areas. In 1986 the Vermont Legislature addressed the problem by passing Act 199, an Amendment to 10 V.S.A. Chapter 47. Act 199 placed large (greater than 6,500 gpd) nonpoint (land-based) wastewater disposal systems under strict regulatory guidelines, in the form of "indirect discharge permits," to prevent their groundwater discharges from "significantly altering" the aquatic biota of the adjacent receiving streams. The macroinvertebrate community is being used by the Vermont Department of Environmental Conservation (DEC) as a tool to determine compliance/noncompliance under the indirect discharge permit program. Four macroinvertebrate metrics are presently being used to measure the biological significance of changes to the aquatic ecosystems in high quality upland streams. The metrics - Pinkham Pearson Coefficient of Similarity (PPCS); Density; Ephemeroptera, Plecoptera, Trichoptera (EPT);, and a Vermont modified Hilsenhoff Biotic Index, are used to measure changes in the structure and function of control compared to evaluation sites on the monitored stream.

Introduction

Act 199, a law passed by the Vermont legislature during its 1985-86 session, seeks to prevent ultra-oligotrophic upland streams from being altered and enriched via permitted groundwater discharges from large land-based wastewater disposal systems. The upland streams of the Green Mountains typically contain very low concentrations of nutrients and minerals with concentrations of total phosphorus often less than 0.015 and alkalinities less than 10 mg/l (Scott 1983, unpublished data DEC). As a result these streams are very sensitive to change, even from slight increases in nutrients, minerals, metals and acidity from both surface water runoff and their groundwater aquifers. These chemical changes can be due to land-based wastewater disposal systems, altered watershed land use activities, and from acid rain.

A typical undisturbed drainage often contains shallow soils, is forested, and has steep gradient, forming streams with substrates that are dominated by boulder, rubble, and gravel. The forest canopy is usually complete, which limits light penetration and maintains cool temperatures year round. These conditions typically result in aquatic communities that tend to be heterotrophic with comparatively low instream primary production. Diatoms and mosses tend to dominate the periphyton communities. The macroinvertebrate communities are comparatively low in total species richness and density, and proportionally high in Ephemeroptera, Plecoptera, and Trichoptera species richness and density. Overall, the macroinvertebrate communities are generally high in species diversity. Fish communities are often represented by low densities of Brook Trout and Slimy Sculpin.

Characteristics of the aquatic macroinvertebrate communities at spatial control and test sites are being used to evaluate changes to the biological integrity of these ultra-oligotrophic upland stream communities caused by groundwater discharges from nonpoint regulated wastewater disposal sites. The VDEC favors the use of the macroinvertebrate community over the other aquatic communities as a compliance monitoring tool for the following reasons: (a) the macroinvertebrate community mirrors changes at the primary trophic level, (b) the macroinvertebrate community is more complex and diverse in small streams than the fish community, and (c) macroinvertebrate communities are found in all types of aquatic habitats, thus allowing for instream biomonitoring to take place even in the smallest high elevation mountain brooks. In addition, the VDEC feels that the following sampling and analysis considerations favor the use of the macroinvertebrate community: (a) macroinvertebrate sampling procedures can be adapted to all habitat types and standardized to a sufficient degree to allow for comparable data generation by different dischargers, (b) the use of artificial substrates often allows for better evaluation of community changes in paired site comparisons, and (c) sampling and analysis procedures allow for quantitative sampling and quality assurance procedures to be developed so that metric variations and the precision and accuracy of the data base can be evaluated.

In order for the macroinvertebrate community assessments to become part of the regulatory program, the assessment protocols must be performed by the discharger and be standardized containing quality assurance checks. Assessments must be cost efficient. Representative samples must reflect the instream community being monitored and optimally be precise. Assessments must have standardize data analysis and sample processing protocols, including the level of taxonomy performed and the metrics used to demonstrate changes in the biological integrity of the stream community. Assessments must create a vehicle by which biocriteria can be incorporated into the regulatory process so that reasonable decisions on the extent of compliance can be made.

Methods and Materials

A project plan is required by the VDEC prior to the issuance of the State nonpoint discharge permit. The plan must describe in detail how the biomonitoring will take place, personnel involved, detailed site descriptions, sample handling and reporting protocol, and quality assurance procedures used in the field and laboratory. The plan must follow all VDEC recommended protocols (VDEC 1987) or justify any deviation. Review and approval of the plan is necessary by biologists prior to the initiation of monitoring.

The discharger is responsible for sample collection and processing, and a final report. The VDEC biologists are responsible for data and metric review, analysis of the site, and compliance decisions. All work must be done by an aquatic biologist or by personnel under the supervision of an aquatic biologist. The supervising biologist must have considerable background or academic coursework in aquatic macroinvertebrate ecology. The qualifications of the personnel involved are included in the project plan.

The primary monitoring strategy is the paired site comparison with no temporal controls (Green 1979). Stream sites above and below the zone of groundwater influence are compared to determine compliance. If an upstream control site is unavailable, an appropriate ecoregional control site is selected from adjacent streams in the drainage on a site by site basis. The site selection objective is to isolate the discharge as the only potential disturbance between the control and test sites and to minimize intersite physical habitat and longitudinal biological variability.

Sampling takes place during the months of August to mid-October. This time period generally coincides with the anticipated seasonal low flow and high temperature period, when the contribution and influence on a stream's chemistry by its groundwater aquifers is greatest.

It is essential to use a sampling method that will both provide a representative sample of the stream biota and is capable of determining the compliance criteria. The recommended sampling device is a rock-filled (lo-boy) basket with minimal dimensions as described in the VDEC protocols. This sampling device allows for quantitative data generation and optimizes the precision of the metrics by reducing substrate related variability between sites. Precision of the VDEC metrics using five replicates generally range as follows: density 10 to 30% standard error; ephemeroptera, plecoptera, and tricoptera (EPT) 6 to 12% standard error; and biotic index 3 to 6% standard error of the mean.

In addition to the quantitative sampling, a qualitative D-net kick sample is collected at each site. The VDEC rapid assessment protocol is used (VDEC 1987). This sample is submitted to the VDEC before laboratory processing and is utilized as a quality assurance taxonomy check and a measure of the gross compositional representation of the quantitative sampling technique. The information is also being used in developing an ecoregion expectation for establishing water quality biocriteria.

Field observations are useful in the regulatory process when recorded on standardized forms. This insures that the sites habitat are routinely observed, categorized, and documented. The field observations can then be clearly interpreted facilitating effective use in the regulatory process. Two field sheets are used per site, one documenting the stream habitat characteristics and the other documenting area specific placement of the rock basket substrates.

Sample processing is done in a laboratory. Taxonomic identification is made to the lowest possible level consistent between sites. It is essential that the level of identification be standardized within the program since data is incorporated into biological metrics of community integrity to measure degrees of biological change between sites.

Results and Discussion

<u>Metrics</u> -- The data generated by

these methods are used to calculate four metrics. The degree of change between the control and test site is used to determine the significance of the alteration to the aquatic biota of the stream. The overall magnitude of the change will be used in determining an appropriate regulatory action.

Metric 1 -- Pinkham Pearson Coefficient of Similarity (PPCS) Index --The PPCS (1976) is used to make an immediate compliance decision when alteration of the aquatic biota is extreme. The index measures the magnitude of intermediate effects. The PPCS involves a stepwise comparison of abundance and structural factors, providing a numerical indicator of changes in the standing crop and taxonomic structure (biological integrity) of the macroinvertebrate community from control to impact area. Values range from zero indicating total dissimilarity to one indicating total similarity. The PPCS is calculated as follows:

$$B = \frac{1}{K}\sum_{i=1}^{K} \frac{\min(xia,xib)}{\max(xia,xib)}$$

where: B = Coefficient of similarity
K = the number of comparisons between stations
xi = the number of individuals in taxon i
a,b = site a, site b

The PPCS is calculated using the major taxonomic components of the macroinvertebrate community. A major taxonomic component is a generic level (or higher) taxonomic grouping of organisms which compose greater than 3.5% numerical density of the community. The mean numerical density of each major taxonomic component of both the control and evaluation sites are calculated. The major component modification of the PPCS Index is being applied because the index is the mean of the summed proportional quotients of the values entered. Since macroinvertebrate communities often contain many rare taxa, loss from the community creates zero quotients which results in a skewed rating of the rare taxa loss compared to density shifts of dominant taxa (Brock 1977). The absence of rare taxa from a site list may only be due to the quantitative sampling methods (Allen 1975). By only using major taxonomic components of the community the index is a reflection of changes in the overall integrity between the two communities.

PPCS values greater than 0.75 result as a function of natural spatial variability in macroinvertebrate distribution, as well as sampling error. PPCS values in the range of 0.25 to 0.75 indicate a possibility of a significant alteration in the biological integrity of the aquatic macroinvertebrate community has occurred. PPCS values less than 0.25 indicate a significant alteration to the biological integrity of the aquatic macroinvertebrate community. Values in this range indicate a large change in abundance or a major restructuring of the macroinvertebrate community. Which may include the reduction or elimination of major taxonomic orders or the reduction and elimination of functional processes within the macroinvertebrate community. Alterations of this magnitude may indicate profound effects on the entire aquatic ecosystem.

Table 1. Hilsenhoff Biotic Index rating for assessing water quality based on benthic macroinvertebrate collections.

Biotic Index Value Range	Water Quality Rating
BI < 2.0	Excellent, Undisturbed
2.0 < BI < 2.5	Good
2.5 < BI < 3.0	Fair
3.0 < BI < 3.5	Poor
BI > 3.5	Very Poor

<u>Metric 2 -- Biotic Index</u> -- Hilsenhoff Biotic Index (BI) (Hilsenhoff 1982), with specific modifications made by the VDEC for the aquatic macroinvertebrate fauna of Vermont (VDEC 1987) utilizes the indicator organism concept. Each taxa is assigned a tolerance value based on response to nutrient enrichment. As with the PPCS, the BI is an integrating index which utilizes information concerning both the relative abundance and organic pollution tolerance of individual taxa. The evaluation involves the analysis of each taxons pollution tolerance to relative abundance in the community. The following formula is used to calculate the BI:

$$BI = \Sigma \frac{(ni \times ti)}{N}$$

Where: N = total number of individuals in sample;
n_i = number of individuals of taxon i;
t_i = tolerance value assigned taxon i.

Tolerance values for individual organisms and the community range from 0 to 5 (0 = intolerant, 5 = tolerant), as well as, the community BI (0 = pristine, 5 = extremely degraded). Biotic index values of zero or five are unlikely to occur, most often values range between one and four. The driving factor for the low to mid-range BI values is related to changes in the macroinvertebrate food base, consisting primarily of periphyton and particulate organic material. Values at the upper end of the scale are associated with extreme organic degradation (i.e., high BOD and NH_3, low dissolved oxygen).

Values for the BI follow that of Hilsenhoff (1982) with modification based on VDEC data for interpreting water quality (Table 1).

The following conclusions can be drawn: pristine streams in the Green Mountains of Vermont typically have BI values less than 2.0. As more fine particulate organic matter and periphytic algae become available the community BI value increases. This increases the BI value from less than 2.0 to greater than 3.0 depending on the

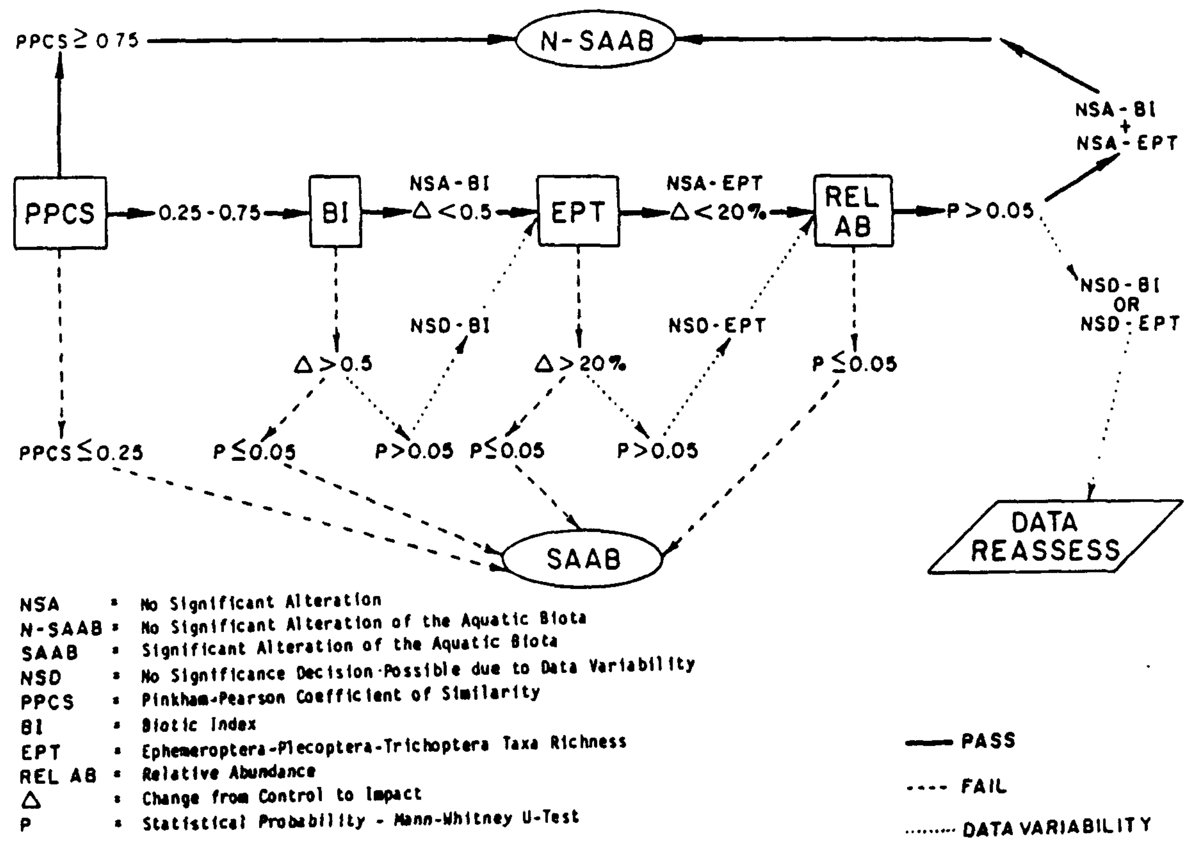

Fig. 1. The macroinvertebrate biometric series and criteria used to determine compliance with the significant alteration of the aquatic biota standard in Vermont.

magnitude of food increase. The mean statistically significant (P < 0.05 Mann-Whitney U-Test) change in BI is 0.3 units, thus a change greater than 0.5 indicates a significant alteration of the aquatic biota. An increase indicates an enrichment impact while a decrease indicates a depressive influence on productivity in the receiving water.

Metric 3 -- Ephemeroptera, Plecoptera, Tricoptera (EPT) Index -- The EPT index is a sensitive indicator index since mayflies, stoneflies, and caddisflies are a major proportion of most lotic aquatic macroinvertebrate communities in Vermont. Changes in the taxa richness and diversity of this subcommunity provides a reliable indication of changes to the biotic integrity of the aquatic biota (Penrose et al. 1980; Lenat et al. unpublished data; Bode unpublished data). The EPT index is calculated as follows:

$$EPT = \frac{\sum_{i=1}^{N} EPTi}{N}$$

Where: EPTi = number of EPT taxa from replicate i

N = number of replicates

The EPT metric measures alterations to the integrity of the aquatic biota from organic enrichment (enhancement and degradation), toxicity, and habitat degradation (Lenat 1983). Taxa-poor headwater streams will increase in taxa richness as more food sources become available (Vannote et al. 1980). Decreasing taxa richness has been documented in cases of iron precipitate coated stream beds, toxic dissolved metals from old copper mine drainages, and siltation and substrate embeddedness from development activities.

The EPT index criteria presently used to indicate a significant alteration to the aquatic biota is a 20% change from control to evaluation site. If the EPT metric is statistically significant ($p < 0.05$ two-tailed Mann-Whitney U-test) using five replicates per site. If the data set is statistically inconclusive due to low control richness values or high data variability total EPT taxa or changes in the percent composition of the EPT orders are evaluated.

Metric 4 -- Density Metric -- Changes in the density of aquatic macroinvertebrates evaluates changes in gross primary and secondary biological production. Density is the number of organisms per replicate from a sampling area. Estimates of density are relatively imprecise (SE ranges between 10 to 40%), because of this any statistically significant change in density from control to impact area presently constitutes an alteration of the aquatic biota. A decrease in density of 50% or increase in density of 150% should be considered biologically significant. A decrease in density criteria of 30 to 60% has been considered a moderate stress by other researchers (Penrose et al. 1980), and is presently being recommended as a biologically significant alteration of the aquatic biota. The Mann-Whitney U-test is applied to test statistical significance of observed changes at $p < 0.05$.

Regulatory Evaluation

A series of comparative metric analyses will determine the severity of alterations to the aquatic biota. Exceedance of a single metric constitutes a significant alteration of the aquatic biota outside the bounds of natural variability. The process by which the "significant alteration" criteria is evaluated is detailed in Fig. 1. In order to establish compliance all four parameter values must be within the no significant alteration range. If the change in any one parameter exceeds the criterion the discharger will be out of compliance.

Regulatory Actions

The severity of regulatory response is based on the magnitude of alteration to the aquatic biota, the best professional judgment of the VDEC biologist, and the degree site-specific conditions indicate the groundwater effluent was the primary cause. The amount of change between the control and evaluation sites for the metrics out of compliance and the number of concurring exceedances determines the magnitude of alteration.

The VDEC regulatory response has ranged from forced abandonment of a leachfield, correction of treatment facility operating failures, to increased biomonitoring requirements in a discharge permit.

Compliance biomonitoring located in rapidly developing resort areas have documented stream sedimentation problems which were unrelated to the effects from the groundwater discharges. The biosurvey data was able to document the degradation on the stream, and focused the attention of the VDEC and the developer on the problem at the resort.

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RAPID BIOASSESSMENTS OF LOTIC MACROINVERTEBRATE COMMUNITIES: BIOCRITERIA DEVELOPMENT

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Abstract

Traditionally, the examination of resident biota has been recognized as perhaps the most straightforward method of assessing water quality since conditions must be favorable for a balanced biological community to exist and perpetuate. Biosurveys are an important method of identifying impairment of aquatic life and can easily be used in conjunction with other biological and chemical monitoring tools in the design of biocriteria. However, from the regulatory standpoint, biological monitoring has had its share of shortcomings. For statewide monitoring programs, the classical intensive quantitative evaluations of biotic communities have been, in many cases, too labor-intensive, time-consuming and expensive. Often, the usefulness of the data has been limited since only aquatic ecologists could understand it.

The increased emphasis on the receiving stream and water quality-based limits created a need for the development of abbreviated methods of generating useful biological data. In the early 1980's, aquatic biologists produced rapid bioassessment techniques and provided information on the concept at the 1986, 1987 and 1988 annual meetings of the North American Benthological Society. Further development of these techniques has continued by numerous state agencies and at the federal level with EPA providing technical guidance (Plafkin et al. 1987). The realization that rapid bioassessments can overcome previously ineffective applications of biological methods is gaining acceptance in the water quality management community. Impact assessment information can now be readily obtained in a cost-effective manner. Rapid bioassessments are useful for screening and as a good starting point when an integration of methods is appropriate.

The primary objective of this report is to convey information pertaining to the validity and reproducibility of a rapid bioassessment technique initiated by the Biomonitoring Section of the Arkansas Department of Pollution Control and Ecology (ADPCE) in 1986. A pilot study was conducted whereby comparisons were made between the complete laboratory analysis of a five-minute riffle samples and field processed 100-organisms rapid bioassessments. Investigator subjectivity was tested through a sampling regime of replicate samples collected at: 1). the same riffle by the same individual, 2). the same riffle by two different individuals, 3). two successive riffles in a minimally stressed stream by the same individual and 4). two successive riffles in a minimally stressed stream by two different individuals. Examples of the data generated from these methods are included in this report. A scoring system, using biometrics, was designed to include

qualitative and semi-quantitative measures of the aquatic macroinvertebrate community to develop biocriteria for determining aquatic life use status. The biometric scoring criteria were structured from data generated by the replicate samples which revealed variations between any two samples taken at the same site.

Various levels of uncertainty have been encountered in the application of numeric criteria due to the complexity of aquatic ecosystems. In some scenarios the so-called "safe number" may not adequately protect aquatic life, while in others, unnecessary regulatory requirements prevail. This does not imply that numeric criteria have no place as a management tool, but their application may be enhanced when supplemented with narrative biological criteria developed from biosurveys of ambient fauna. There is no better way to determine the aquatic life use status of a stream than to examine its inhabitants.

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SEMIQUALITATIVE COLLECTION TECHNIQUES FOR BENTHIC MACROINVERTEBRATES: USES FOR WATER POLLUTION ASSESSMENT IN NORTH CAROLINA

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Abstract

Semiqualitative collection methods for benthic macroinvertebrates have been used in many types of water pollution studies in North Carolina. These techniques emphasize multihabitat collections and the use of both coarse and fine-meshed samples. These techniques have proven to be more cost-effective and have generated more useful data than conventional collection techniques. Examples of how data are used in several monitoring programs are presented and include trend monitoring, point source surveys (including toxicity reduction) and use attainability.

Introduction

Benthic macroinvertebrates have been collected using a variety of quantitative and qualitative techniques by many state and federal agencies. Quantitative approaches (i.e. Hester Dendy multiplates or Surber samples) are thought to be more precise and more amenable to statistical analysis; however, quantitative techniques are both time and cost intensive. Quantitative sampling is also usually habitat specific, resulting in a large portion of the aquatic community not being sampled. For example, Allan (1975) found that twelve Surber samples underestimated total taxa richness (from a variety of collection methods) by about 32%.

The North Carolina Division of Environmental Management (DEM) uses aquatic macroinvertebrates to assess water quality in streams and rivers. A large number of sites are sampled each year (300+) and the information gathered is used to document both spatial and temporal changes in water quality. To deal with this large volume of work, a new semiqualitative collection technique was developed. This technique samples most microhabitats using both coarse-mesh and fine-mesh samplers. Abundance values are qualitative (rare, common, abundant), but taxa richness information is quantitative.

North Carolina Monitoring Programs

The North Carolina Division of Environmental Management (DEM) conducts several types of water quality assessment programs. Emphasis has been directed towards measuring the effects of point source dischargers (and toxicity reduction) and trend (or ambient) monitoring. Water quality surveys are also conducted to assess non-point source pollution, water use attainment and spill assessment. The use of semiqualitative collection techniques for benthic macroinvertebrates has allowed biological monitoring data to be incorporated into each of these monitoring programs.

Semiqualitative Method Description

The method description presented here is an excerpt from a paper (in preparation) written by David Lenat. This paper provides a complete discussion of North

Carolina's qualitative collection technique.

This technique is intended for use only in wadable, freshwater streams. Sampling is easiest during periods of dry weather, as high flow conditions may severely impair sampling efficiency by making critical habitats inaccessible. Collections are designed to have a fixed number of samples at each site (10), although many of these are composite samples. Six different collection methods are utilized to collect qualitative samples from a variety of microhabitats. All samples are picked in the field.

Kick nets: Two kick samples are taken, usually in riffle areas. In very small streams, or in sandy areas without riffles, kicks are taken from root masses, "snags", or bank areas. All types of benthic macroinvertebrates can be collected with this sampling technique, but emphasis is placed on the collection of Ephemeroptera, Plecoptera and Trichoptera. Kick net samples are washed down in a large bucket sieve prior to being processed.

Sweep nets: Three samples are taken by physically disrupting an area and then vigorously sweeping through the disturbed area with a long-handled triangular net (approximately 1 mm mesh size). Sweeps are usually taken from bank areas and/or macrophyte beds. Bank areas are particularly important for the collection of species which prefer low current environments. In particular, samples are inspected for Chironomini, Oligochaeta, Odonata, mobile cased Trichoptera, Hemiptera, Sialis, Crustacea, and Ephemeroptera. Large rocks or bedrock areas with attached macrophytes (especially river weed, Podostemum) also may be sampled with the sweep net to look for Hydropsychidae, Baetidae and Ephemerellidae. A sweep net can also be used to collect gravel, which is inspected for stone-cased Trichoptera. The latter technique is used principally in the sandhills area of North Carolina, an ecoregion with a diverse trichopteran assemblage.

Fine-mesh sampler: Smaller organisms, especially Chironomidae, are sampled with a finer mesh (300 microns) and are field-preserved to increase picking efficiency. Rocks or logs with visible growths of periphyton, Podostemum or moss are washed into a large plastic basin (or bucket) partially filled with water, and the substrate is vigorously brushed or rubbed to dislodge all attached fauna. Any large particulate material (leaves, etc.) is washed and discarded. A single composite sample is made from several rocks and logs. The material remaining in the basin is poured through the fine mesh sampler, which is constructed of 4 inch PVC, and the water drained completely. The residue is quickly preserved in 95% alcohol by placing the PVC cylinder into another slightly larger container half filled with 95% alcohol. The sample soaks in alcohol for about five minutes, and then is backwashed with stream water into a picking tray. This method of field preservation requires only a small amount of reusable alcohol.

Sand samples: Sand habitats often contain a very distinct fauna, but extraction of this fauna, using dredges, cores, etc., can be very tedious. Sand substrates are sampled with a large bag constructed of fine mesh (300 microns) nitex netting. It can be quickly constructed from a one meter square piece of netting,

Table 1. Taxa richness criteria (10 samples) for assigning water quality classification to free-flowing, wadable North Carolina streams and rivers, July to September.

Class	EPT* Taxa Richness Mountain	Piedmont	Coastal	Total Taxa Richness Mountain	Piedmont	Coastal
Excellent	>41	>31	>27	>91	>91	>83
Good	32-41	24-31	21-27	77-91	77-91	68-83
Good-Fair	22-31	16-23	14-20	61-76	61-76	52-67
Fair	12-21	8-15	7-13	46-60	46-60	35-51
Poor	0-11	0-7	0-6	0-45	0-45	0-35

* Most intolerant groups = Ephemeroptera + Plecoptera + Trichoptera

folded in half and sewn together on the opposite side and the bottom. This bag is used like a Surber sampler, but the lack of a rigid frame allows for easy storage when folded. The bag is held (open) near the substrate, and the sand just upstream is vigorously disturbed. A composite sample is collected, utilizing 2 to 3 locations. The material collected (a lot of sand and a few organisms) is emptied into a large plastic container half-filled with water. A "stir and pour" elutriation technique is used in conjunction with the fine mesh sampler described above. After field preservation, the elutriate is checked for Chironomidae (especially <u>Rheosmittia</u>, <u>Harnischia</u> group and <u>Polypedilum</u> spp.), Oligochaeta, Gomphidae and some Ephemeroptera.

<u>Leaf-pack samples</u>: A large, coarse-meshed bucket sieve is used to wash down "aged" (decomposing) leaf-packs, sticks and small logs. Such samples are particularly helpful in large sandy rivers where many of the species are confined to "snags" (Benke et al. 1984, Neuswanger et al. 1982). This technique is good for finding "shredders", especially Tipulidae, Plecoptera and Trichoptera.

<u>Visual search</u>: Large rocks and logs are visually inspected for any associated invertebrates. Certain tightly adhering organisms may be collected only by this technique (Lepidoptera, Blephariceridae, <u>Leucotrichia</u>, <u>Psychomyia</u>). Decaying logs are picked apart, especially logs with loose bark. Freshwater sponges are inspected for Chironomidae (<u>Xenochironomus</u>), Trichoptera (<u>Ceraclea</u>) and Neuroptera. Rocks near the shore (in negligible current) will harbor certain Ephemeroptera and Odonata, and leaves near the shore may be primary habitat for some Gastropoda. In addition, a mussel search is conducted by careful visual inspection of the bottom.

<u>Sample processing/identification</u>: A standardized qualitative

collection consists of ten samples: two kicks, three sweeps, three fine mesh samples (two rock-log samples and one sand), one leaf-pack and visuals (considered as one sample). All samples are field picked with jeweler's forceps from white plastic or enamel trays and preserved in 6 dram vials. Organisms are picked in proportion to their abundance, but no attempt is made to remove all organisms. If an organism can be reliably identified as a single taxon in the field, no more than 10 individuals need to be collected.

Samples are identified in the lab, using species level identification when possible. Chironomidae and Oligochaeta are sorted under a dissecting microscope and representative individuals are slide mounted. During sample processing and identification, it is fairly simple to add some measures of abundance. As invertebrates are identified, they are recorded as abundant (>9), common (3-9), or rare (<3). Field notes can also be used to label exceptionally abundant (dominant) species. Total taxa richness and taxa richness for Ephemeroptera, Plecoptera and Trichoptera (EPT) are calculated and used to assign a biological classification to each station (see criteria development).

EPT surveys: A general assessment of water quality can also be obtained quickly using an abreviated method which involves collecting only four samples (1 kick, 1 sweep, 1 leaf-pack and visuals) and identifying only Ephemeroptera, Plecoptera and Trichoptera. This technique, called an EPT survey, is used to supplement full qualitative surveys at a greater number of locations. The information can be used to assign relative water quality classifications and determine if additional or supplemental surveys should be conducted. In some instances the EPT survey may be used to assess impacts to the intolerant EPT groups. At this point, only preliminary comparisons have been made between EPT values collected during full qualitative (10 samples) and EPT surveys (4 samples). These comparisons (N=10) suggest that full surveys collect 1.1 to 1.3 times more EPT taxa. More testing in specific ecoregions and stream orders need to be conducted.

Criteria Development

Criteria have been developed for three major geographic regions (NC DEM 1986, Table 1) to relate both total taxa richness and EPT taxa richness to five water quality classifications: Excellent, Good, Good-Fair, Fair and Poor. These criteria have been used since 1983 with little modification. Upper (Excellent classification) and lower (Poor classification) limits were established based on collections in each region at both unstressed and highly polluted sites. The other three classes were then defined by dividing the remaining taxa richness ranges into three equal groupings.

Data collected from unimpacted sites suggest that there is a relationship between region and the taxa richness of the benthic macroinvertebrate community. This assumption was tested by comparing average taxa richness data from summer collections (July to September) at unstressed sites in the mountains (7 sites), the piedmont (6 sites) and the inner coastal plain (4 sites). Areas that had intermediate characteristics (upper piedmont) were not included in this analysis. The majority of the sites had been sampled in more than one year (up to six years) and average taxa richness values were utilized. "Unstressed sites" were defined as areas with no known chemical/physical alterations, having a high diversity of invertebrates and/or fish. Going east from the mountains to the coastal plain, taxa richness decreased for Ephemeroptera (20.6 , 15.8 and 10.1, respectively) and increased for Odonata (4.6, 8.2 and 11.1, respectively). Plecoptera, Trichoptera and Diptera were most diverse in the mountains, while Coleoptera, Crustacea and Mollusca were more diverse in the piedmont/coastal plain areas. The EPT subtotal was clearly higher in the mountains (44.7 as compared to 31.9 in the piedmont and 28.7 in the coastal plain), but total taxa richness differences were not as great (mountain = 98.9, piedmont = 93.3 and coastal plain = 91.0).

Water Quality Assessments Using Benthic Macroinvertebrates

Semiqualitative collection techniques have been used successfully in a number of monitoring programs. The most ambitious program in North Carolina is the Benthic Macroinvertebrate Ambient Network (BMAN). However, these collection methods are flexible enough to allow for the use of benthic macroinvertebrate data in a number of other programs. Several of these include point source monitoring (including toxicity reduction), use attainability analyses and water use classification.

Benthic Macroinvertebrate Ambient Network (BMAN)

The North Carolina Division of Environmental Management (DEM) maintains 350 ambient locations in seventeen (17) major river basins. Water quality data is collected from each site by personnel in 7 regional offices on a monthly or quarterly basis. Benthic macroinvertebrate samples have been collected from a total of 208 of these locations to assess long term trends (since 1982) in water quality. Samples are collected from 80-100 of these sites each year. These locations are staggered each year (collections are made either on an annual, biennial or triennial basis so that greater state wide coverage is feasible). BMAN samples are collected during summer months to lessen temporal variation between years and document worst-case (low flow, high temperature) conditions. Between-year changes in the composition of the benthic community structure, including differences in total taxa richness (ST and SEPT, and subsequent bioclassification) as well as the occurrence of dominant taxa or "indicator groups" are used in the analyses. Between-year analyses also include important flow related variables. These include changes in current velocity, changes in non-point contributions and changes in the length of recovery zones below point sources.

Biological data are tabulated each year summarizing trends in water quality. In 1986 for example, positive trends were noted at eighteen (18) BMAN locations while negative between year trends were noted at only four (4) stations. These data were collected during an extremely low flow period in North Carolina, which probably tended to reduce the effects of non-point sources of pollution. The effects of point sources also may have been increased at some sites by reducing effluent dilution. Additionally,

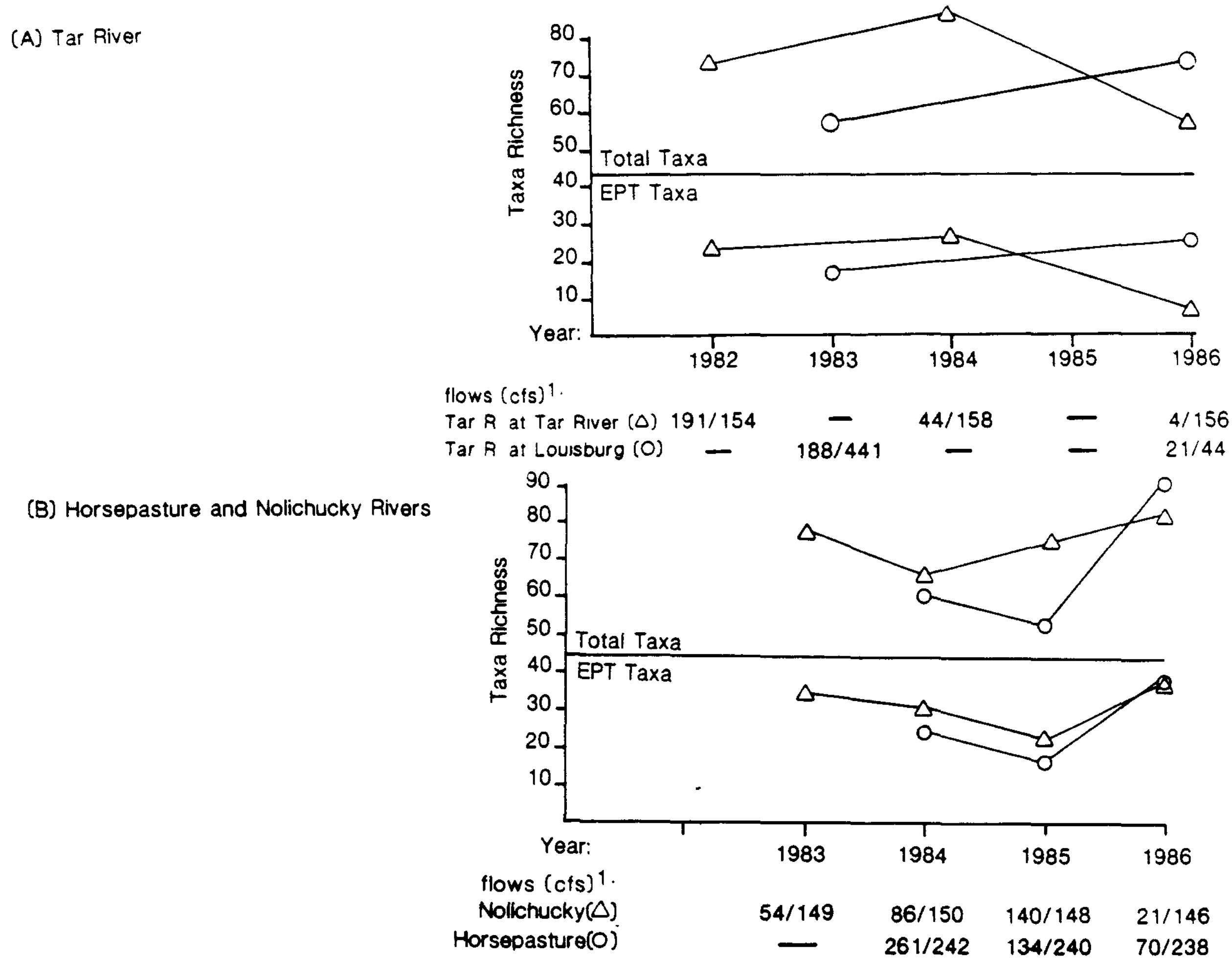
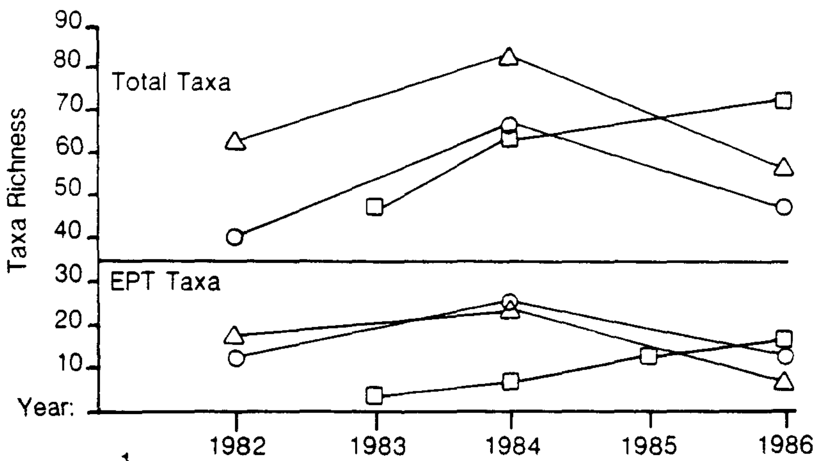


Fig. 1. Trend analysis effects of flow and non-point source impact on benthic taxa richness.

between-year changes in benthic fauna are compared to changes in chemical water quality data.

Biological data have been useful in documenting between-year changes in water quality, although interpretation can not be limited to biological data. For illustration, several examples are provided. Figure 1A and 1B illustrate taxa richness data and flow at several stations from watersheds having significant non point source (sedimentation) contributions. Data from 1986 indicate that reduced flows significantly reduced taxa richness from the Tar River at Tar River site (Fig. 1A) because flows were almost negligible (3% of the average flow), thus reducing the number of taxa dependant on flow. However, taxa richness values were greater at the downstream Tar River at Louisburg site in 1986 than in 1983. Flow, although reduced at Louisburg in 1986, was sufficient enough to maintain flow-dependant taxa (i.e. filter-feeders), but reduced the effects of non point source pollutants. Low flow conditions improved taxa richness and, in this case, increased the bioclassification from Good-Fair to Good.



flows(cfs)[1]:	1982	1983	1984	1985	1986
Lower Little River (△)	530/417	—	1056/427	—	26/415
So.Fk. Catawba (O)	33/90	—	100/91	—	6/89
Deep River (□)	—	43/357	230/357	402/355	131/354

[1] Flows are expressed as f30/average flow = average flow 30 days prior to benthos collection/average flow for period of record.

Fig. 2. Trend analysis effects of flow and point source impacts to benthic taxa richness Lower Little, South Fork Catawba and Deep Rivers.

Data are illustrated from two mountain rivers (the Horsepasture and Nolichucky Rivers) in western North Carolina in Figure 1B. These data indicate that during high flow years (1984 and 1985) non-point source contributions are greater, thus reducing taxa richness values. However, taxa richness values increased in 1986 during extremely low flow conditions. Much of the Horsepasture River watershed is currently being developed for second homes and tourism and therefore subject to impacts from sedimentation. Continued monitoring should detect the effect of these activities on the instream fauna.

Figure 2 illustrates data from stations selected below major municipal dischargers. In 1984, during high flow, the benefits of dilution to instream fauna are evidenced at both the Lower Little and South Fork Catawba River locations. Instream waste concentrations were greater during low flow years (1982 and 1986) and taxa richness values were lower. The exception to these trends are data from the Deep River. Facility upgrades have improved water quality in the basin (NRCD 1988a), resulting in increased taxa richness and improved water quality even during low flow periods with less dilution of impacts.

In 1984, facility upgrades (denitrification) were also completed on the major municipal above the South Fork Catawba River station. Lower ammonia levels and greater flow (dilution) in 1984 resulted in increased taxa richness. Also, many toxic tolerant chironomids (<u>Cricotopus</u> <u>bicinctus</u>,

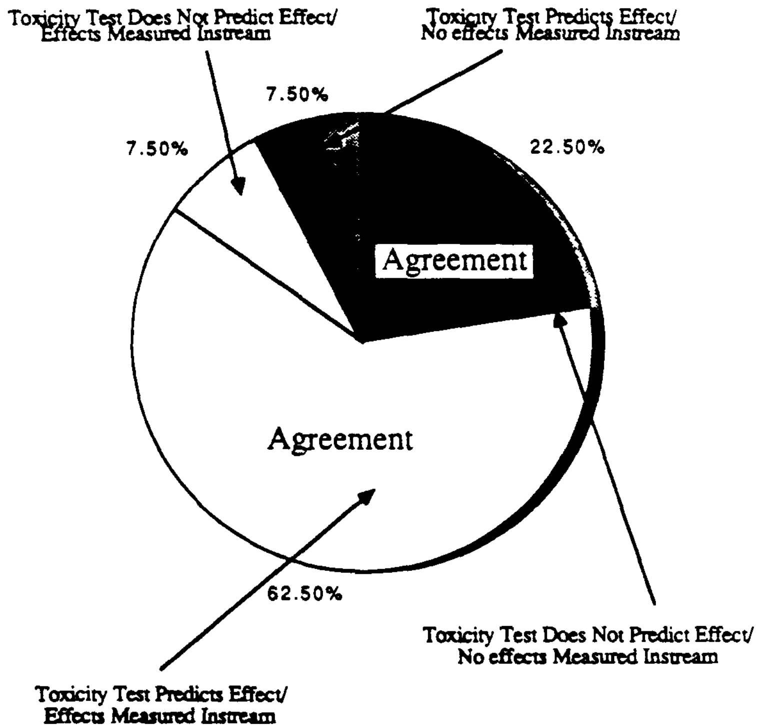


Fig. 3. Whole effluent toxicity measurements and resultant instream impacts as reflected by benthic macroinvertebrate populations.

C. tremulus, C. varipes and Polypedilum illinoense) were abundant in 1982, but absent or reduced in abundance in 1984. However, taxa richness was again reduced in 1986 during low flow and the toxic tolerant chironomids were replaced by enrichment indicators (Chrionomus sp., Tribelos sp., Tanytarsus sp. and Limnodrilus hoffmeisteri). These latter observations indicate the value of using "indicator groups" in trend analysis.

Point Source Monitoring Programs

The primary responsibility of many state biological monitoring programs is to assess point source impacts. Simply assessing the effects of a point source using the indigenous aquatic fauna can be achieved using several techniques. However, qualitative collection techniques have allowed biological data in North Carolina to be used effectively to direct and/or supplement management strategies. For example, benthic macroinvertebrate data is used as a screening tool to occasionally direct more labor intensive chronic toxicity studies, to supplement effluent toxicity testing, to assess the effectiveness of facility upgrades and also to supplement water quality investigations of fish kills or other episodic events detrimental to water quality.

Currently the use of instream benthic surveys to test for toxicity are not required in the NPDES permitting process; however, benthic data has been collected to supplement whole effluent toxicity testing at 40 North Carolina facilities. This information is used to identify impacts not addressed by numerical standards. Figure 3 illustrates strong agreement (85%) between results of chronic toxicity testing using Ceriodaphnia which predict instream toxicity and benthic macroinvertebrate surveys which have measured instream toxicity (Eagleson et al. 1988). Only in six of the 40 surveys (15%) did effluent tests and benthos surveys not agree. In two of the three instances where benthos surveys did not detect instream toxicity predicted by assay, poor upstream water quality was noted which masked any downstream toxicity.

One goal of the 1972 amendments to the Federal Water Pollution Control Act was to require secondary effluent limits for all wastewater treatment plants. However, this requirement resulted in a great deal of debate over whether or not meeting secondary effluent limits (at a considerable expense to municipalities) would result in better water quality. To test the effectiveness of these additional controls, seven biological surveys were conducted (and used to supplement chemical surveys) before and after facility upgrades (NRCD 1984). The results of these investigations indicated that moderate to substantial instream improvements were observed at six of the seven facilities. Several more recent investigations also have noted instream improvements following facility modifications and compliance. The use of benthic macroinvertebrate surveys to test for instream improvements following facility modification has proven to be an efficient, cost-effective monitoring tool.

Benthic macroinvertebrate surveys are also used to assess, and identify causes, of fish kills or spill events. If proper protocols are taken, benthic information can be collected and processed rapidly, resulting in the information getting to the enforcement agency often times quicker than results of chemical or fish tissue surveys.

Use Attainability and Water Use Classification

The North Carolina Division of Environmental Management (DEM) has the responsibility of determining water use classifications for all North Carolina surface waters. These uses include water supply, fishing (trout and non-trout), shellfish waters, water contact sports and Outstanding Resource Waters. DEM has the responsibility to assess water use attainment (i.e. are the uses being supported?) and to assess any proposed reclassifications. For example, recent regulations promulgated by EPA (November 1983) require a "use attainability analysis" to be conducted when uses are removed from a stream classification. Use attainability and water use classification involve a comprehensive analysis of physical, chemical and biological factors affecting the attainment of a use. Benthic macroinvertebrate data has played a key role in these analyses and biologists have been principle authors of reports recommending appropriate classifications by evaluating attainable use.

The most recent Water Quality Progress report in North Carolina (NRCD 1988b) indicated that 60.7% of the almost 37,000 miles of freshwater streams and rivers support their intended uses, 24.8% partially support, 4.7% do not

Table 2. Comparison of biology (mostly benthic macroinvertebrate data) vs. chemical data for freshwater stream water use evaluations (NRCD 1988b).

River Basin	Final Evaluation Based on Chemistry	Final Evaluation Based on Biology	Both Biology & Chemistry Data Available: Agree	Disagree: Biology with Lower Rating	Disagree: Biology with Higher Rating
Mountains					
Little Tennessee	16%	84%	70%	0%	30%
French Broad	48%	52%	17%	19%	64%
New	0%	100%	47%	0%	53%
Piedmont					
Broad	28%	72%	88%	0%	12%
Yadkin-Pee Dee	52%	48%	40%	2%	58%
Neuse	42%	58%	58%	0%	42%
Coastal					
Chowan	60%	40%	16%	0%	84%
Lumber	45%	55%	0%	27%	73%
Roanoke	41%	59%	29%	0%	71%
Overall	41%	59%	41%	4%	55%

support and 9.9% were not evaluated. Much of this information was based on biological data and, in particular, taxa richness data from several of the monitoring programs noted earlier. A comparison of how biological and chemical data were used to support uses in several watersheds is outlined in Table 2. Biological data, mostly benthic macroinvertebrates, were used nearly 60% of the time to support the intended use designations. In instances when biology and chemistry disagreed as to a particular use category, 55% of the time biological data were used to assign a higher use support category.

A final water use classification determination in which benthic macroinvertebrate data are used is the designation of Outstanding Resource Waters (ORW). North Carolina's Administrative Code (1986) states that the Environmental Management Commission may classify certain unique and special surface waters of the State as Outstanding Resource Waters (ORW) upon finding that such waters are of exceptional state or national recreational or ecological significance and that the waters have exceptional water quality. This new regulation gives benthic biologists in North Carolina the opportunity to collect samples from unique habitat, to collect rare or unusual taxa and use these and other data sources (including fisheries and fisheries habitat) to recommend appropriate classification.

Special Studies

The use of semiqualitative collection methods to process benthic samples rapidly has allowed our biological monitoring group to assist other federal or state water pollution agencies. Cooperative studies have been conducted with the Forest Service (effects of gypsy moth eradication methods to non-target aquatic insects), the Soil Conservation Service (watershed protection programs and non-point source implementation), the U.S. Geological Survey (acid rain effects on water quality and establishment of a pristine streams network) and local and state councils of governments.

Summary

North Carolina's semiqualitative collection technique for benthic macroinvertebrates was developed to provide a rapid, but reliable assessment of water quality. This technique has proven to be flexible enough to allow biological monitoring data to be used in a number of monitoring programs. These programs include a benthic macroinvertebrate ambient monitoring network which annually summarizes trends in water quality. Ambient data have noted temporal variation in water quality, including flow related variables such as non-point source contributions. Benthic macroinvertebrate data are also used to supplement toxicity testing by measuring instream effects. Effluent toxicity and instream benthic data have an 85% agreement rate and are used by managers to identify impacts not addressed by numerical standards. Semiqualitative collection methods for benthic macroinvertebrates are also used to determine appropriate use classification in use attainability studies in addition to assessing instream improvements due to facility upgrades.

Acknowledgements

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SAMPLING AND DATA EVALUATION REQUIREMENTS FOR FISH AND BENTHIC MACROINVERTEBRATE COMMUNITIES

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Abstract

During the planning of the Workshop, it was decided that the best way to facilitate exchange of the many technical and program/policy-related issues was to establish specific discussion groups of the major issues. From this breakout of groups, two aquatic life discussion groups emerged: benthic macroinvertebrates and fish. The intent of separating these two groups was not to discourage exchange between the two disciplines or establish adversarial positions, but to gather those actively working within the respective aquatic life groups in State and Federal programs for discussion of specific issues. The knowledge and professional experience of a qualified professional field biologist should be used to determine which group of organisms may be best utilized in the various environmental assessment projects. It is also realized that economic constraints sometimes necessitate implementation and operation of a less than ideal program. However, the experiences and knowledge shared at this Workshop convinced us that evaluation of both aquatic life groups is necessary. We <u>do not</u> advocate the use of one group over the other on a programmatic basis. We wish to leave this decision to the State and Federal field biologists.

Introduction

Recommendations from the two workgroups dealt primarily with Quality Assurance. There is a great need to develop and implement Data Quality Objectives, demonstrate reproducibility of results, and ensure that well-qualified biologists (fisheries and benthic) are preforming the sampling and subsequent taxonomic identifications, as well as data analysis and interpretation. Generic and specific recommendations for the benthos and fish community sampling and data evaluation requirements appear below. The following documents provided insight and language for some of the recommendations that appear in this report (Ohio EPA 1987a,b; Hellawell 1978; and Plafkin et al. 1987).

<u>Sampling and Data Evaluation Requirements</u>

The following recommendations apply to both the fish and benthos field programs, thus, they are incorporated together to eliminate redundancy. Bioassessments should be incorporated by the USEPA and State regulatory agencies into an integrated monitoring strategy that can accurately assess instream biological integrity use attainment and measure the success of pollution abatement programs.

The bioassessment program supported by USEPA and the State regulatory agencies should be based on using both fish and benthic macroinvertebrate community analysis. Frequently, analysis of biosurvey sampling results from both aquatic life groups yield the same water quality evaluations and thus corroborates one another.

However, when the results differ, it has been due to the tolerance of these groups to the various environmental stresses that may be reflected only by their trophic level and place in the food chain. Use of the two groups can integrate perturbations to the system via measurements of their structural and functional shifts.

Bioassessments should be used for surveillance, monitoring, and enforcement of water quality standards for point source discharges. Their use identifies synergistic and additive impacts. Small incremental impacts can be identified by quantitative shifts in the community structure and function even when the designated use is being attained.

Bioassessments should be used for screening areas impacted by non-point pollution sources. Once identified, these tools can determine the extent and severity of the impacts and be used to evaluate long-term trends. Results from previous biological surveys should be incorporated into each successive 305(b) report to identify stream segments impact by non-point sources. Whereas there are many surrogate methods for estimating point source impacts to the biota (chemical and bioassay), a direct measurement of the biological communities by biosurveys is the only practical method for assessing non-point source impacts.

<u>Sampling and data evaluation requirements</u>: One sampling event is adequate for screening purposes (Angermeier and Karr 1984) with a single exceedence of a narrative or numerical biocriterion adequate for initiating a control action. Control sites should be selected from upstream sites, for point source impact determination. If the upstream segment is impacted, a control site may be chosen from an adjacent stream of similar drainage area and ecoregion. The ecoregion concept (Larsen et al. 1986; Omernik 1987) may also be useful for watershed evaluation and use attainability categorization when the entire basin may be impacted.

Field sampling must be standardized to ensure reproducibility (OEPA 1987a; Plafkin et al. 1987), and carried out by professional field biologists who determine the appropriate habitats to be sampled and the types of sampling gear necessary. The biologist should participate in all aspects of the field studies from planning, data collection and interpretation, to report writing or at a minimum final document review. Long-term monitoring should be conducted by the discharger to evaluate the effectiveness of the control options.

When evaluating non-point source influences, deficiencies in baseline data should be identified and remediated. This activity should involve EPA interaction with other State, local, and Federal agencies such as U.S. Fish and Wildlife Service, U.S. Forestry Service, U.S. Geological Survey, State DNRs, and DECs. Future resource needs should be coordinated among the agencies to eliminate, or at least reduce, duplication of effort. Other actions include: support of technology transfer between States to enable the establishment of reference control sites for States or EPA regions which have common ecoregions; coordination of basin/watershed studies by the Federal agencies to ensure complete evaluation of transboundary streams; coordination with the State and Regional Superfund groups who may be aware of data necessary to support the non-point source assessments.

If discharger self-monitoring or consultant mediated monitoring is advised by the regulatory agency to support a decision, the following items must be satisfied: the methods used must comply with USEPA or State regulatory agency methods; a study plan must be approved by the regulatory agency; the regulatory agency will determine the level of effort necessary including, but not limited to, the number of study sites, location, taxonomic effort expended; and downstream sampling should include locations within the mixing zone if practicable, and at locations further downstream at discrete intervals to characterize the extent of impact.

Biosurveys should be routinely used to rank grant applications for environmental improvement (remediation) projects based on the potential for environmental improvement. This can be accomplished by comparing the extent and severity of degradation with the potential for biological improvement. The result would be a ranking of where the funds for building or upgrading treatment plants or implementation of Best Management Decisions will result in the greatest environmental benefits for the funds expended.

Fish Community Survey

Fish should be used for environmental assessments since: the taxonomy of fish is well established and allows professional field biologists to identify most species in the field, thus minimizing lab time and speeding data analysis; life histories and environmental tolerances are well documented in the scientific literature; fish comprise the upper trophic levels in aquatic ecosystems thus integrating lower trophic level energy transfer; species specific tolerances to environmental stresses result in measurable shifts in community structure and function; fish continuously inhabit the receiving waters and integrate the chemical, physical, and biological history of the waters that are not directly measured by chemical or short-term bioassays alone; most fish species have long life spans (3-10 years and frequently longer) and can reflect both past and recent environmental quality; assessment techniques now permit determination of the type of impact and incremental degrees using numerical evaluations that have meaning to non-biologists (e.g. Index of Biotic Integrity (Karr et al. 1987), Index of well-being (Gammon et al. 1981), Shannon diversity index, and others used in the State programs); and fish are a highly visible component of the aquatic community to the public sector.

An evaluation of habitat quality must be conducted in conjunction with the biological evaluation to account for ecoregional differences. A standardized quantitative habitat evaluation procedure should be developed.

Although sampling is not limited by season for the purposes of determining environmental impact, sampling should be conducted during the low-flow periods of summer and early fall (Angermeier and Karr

1986; Hilsenhoff 1987). This period generally coincides with the period of greatest environmental stress (e.g. high temperatures, lower DO, lower flow) and ease of sampling.

Larval fish, or young-of-the-year, not used in most indices (Fausch et al. 1984; Karr et al. 1986) should be collected and identified. Their presence should be included in a narrative discussion of the survey results until further indicies can account for impacts to the early life stages.

Sampling should be standardized to obtain a representative sample from each site. Either distance or time needs to be measured, depending on gear type to ensure that a sample is comparable.

The use of ecoregions to delineate and establish reference and control sites is strongly encouraged.

Benthic Macroinvertebrate Community Surveys

Benthic macroinvertebrates provide the impetus for environmental assessments based on the following: most benthos are sessile or have a limited migration pattern, thus they the are particularly well-suited for assessing site-specific impacts (upstream-downstream studies); benthic communities integrate the effects of short-term environmental variations since most species have a complex life cycle of two years or less, the sensitive life stages will respond quickly to stress; degraded stream conditions can often be detected with only a cursory examination of the benthos in the field since they are relatively easy to identify to the family levels for intolerant taxa; sampling is less strenuous than fish, requiring few biologists with inexpensive gear, and has no detrimental effect on the resident biota; benthos are a primary food source for important recreational and commercial fish, and reflect the lower trophic levels; many small streams of 1st and 2nd order naturally support a diverse macroinvertebrate fauna; most State regulatory agencies that routinely collect biosurvey data have benthic data available.

State and Regional training is needed on the use of the Rapid Bioassessment Protocols to further refine and evaluate these methods for geographical-specific uses. The Rapid Bioassessment Protocols should not replace the more extensive State program methods.

State programs represented at this Workshop demonstrated technically successful data generation for supporting water programs. At a minimum, data interpretation should include a decision tree, and should include a range that can be used to evaluate data for the decision process.

The use of artificial or natural substrates should be based on the data needs within a particular State program. Each method provides necessary information for program use and it may be preferable to use the different methods when encountering program or site-specific requirements.

Habitat sampling preference (e.g. riffle, run, pool, undercuts, CPOM) should be decided on a site-specific basis by the State program field biologist. The overall requirement of demonstrating reproducible results negates concern for this item.

Sampling in the mixing zone and downstream of the mixing zone is

Table 1. The comparative ability and "power" of various chemical, physical, and biological assessment techniques to measure or indicate key components of factors affecting biological integrity of surface waters (D - directly measures; I - indirectly measures; S - strongly reflects; C - casual relationship). Modified from Ohio EPA (1987).

Factors/Components	Level 1&2 Exposure Assess.[1]	Level 3 Exposure Assess.[2]	Toxicity (acute)	Toxicity (chronic)	Physical Assessment	Ambient Biological Evaluation
I. CHEMICAL WATER QUALITY						
Conventional substances	D	D	I	I	-	S
Heavy metals	D	D	I	I	-	S
Toxic organics	-	D	I	I	-	S
Static interactions	S	S	I	I	-	N/A
Dynamic interactions	-	I	-	I	-	S
II. ENERGY DYNAMICS						
1° and 2° dynamics	C	I	-	-	-	I
Nutrient cycling	C	I	-	-	-	I
Organic inputs	-	C	-	-	-	I
III. HABITAT QUALITY						
Substrate	-	-	-	-	D	S
Water velocity	-	-	-	-	D	S
Instream cover	-	-	-	-	D	S
Channel integrity	-	-	-	-	D	S
Riparian buffer	-	-	-	-	D	S
Habitat diversity	-	-	-	-	D	S
IV. FLOW REGIME						
Low Extremes	I	I	-	-	-	S
High Extremes	-	-	-	-	-	S
Temporal cycles	-	C	-	-	C	S
Volume	D	D	-	-	D	S
V. BIOTIC RESPONSES						
Acute effects	I	I	D	D	-	S
Chronic effects	I	I	I	S	-	S
Abundance, biomass	-	-	-	-	-	D
Structural	-	-	-	-	-	D
Functional	-	-	-	-	-	D
Disease, etc.	-	-	C	C	-	D
Tolerances	-	-	-	-	-	D
Competition	-	-	-	-	-	S
Reproduction	-	-	-	S	-	S
Predation	-	-	-	-	-	S
Growth	-	C	-	S	-	D

1 primarily models for oxygen demanding substances and simple mass-balance dilution calculations for other substances; steady-state conditions assumed.
2 applications ranging from probabalistic dilution to dynamic fate-assessment models.

recommended to evaluate effluent toxicity and associated impacts.

Benthic invertebrates should be collected during the latter period of the season(s) which demonstrate a stable base-flow (normal flow) and temperature regime. Seasonality and low-flow sampling have been important issues raised by critics of the use of biosurveys in the regulatory programs. We recommend that it is not necessary to sample during 7Q10 conditions because not only do the biological communities integrate the ecosystem effects over a long period of time, the populations are most stable and easily collected at the base-flow allowing for a more accurate assessment.

Benthic surveys are recommended for instances where significant toxicity is measured by effluent toxicity tests. If the biosurvey does not support the toxicity evaluation, the information must be reevaluated using a "weight of evidence" decision (USEPA 1988). If the biosurveys indicate a potential toxicity problem with a discharge, an effluent toxicity test is recommended.

The use of functional feeding groups to indicate use impairment due to toxics or conventional pollutants should be further evaluated, in addition to the utilization of <u>in situ</u> bioassays (e.g. caged fish, clams, leaf-packs) for use in toxicity bioassessment.

The utilization of riparian vegetation as a potential instream assessment tool should also be evaluated.

The EPA biological field methods manual (USEPA 1973) should be updated.

<u>Program Development and Management</u>

Transdisciplinary definitions need to be established for the following terms to eliminate potential confusion over appropriate usage and terminology: biosurvey, biomonitoring, bioassessment, bioassay, biosurveillance, and biotic integrity. Standard definitions may be required for additional terms.

We recommend the use of the term "bioassessment" to broadly encompass the other terms identified. The terms survey (short time period) and surveillance (continued systematic surveys) generally apply to the instream communities (fish and benthos). Toxicity test, or bioassay, are the preferred terms to represent the exposure of test populations in a laboratory setting to ambient water or effluent discharges. <u>In situ</u> toxicity tests, or bioasssays, utilize placement of test organisms in the ambient water or effluent discharges for known exposure periods. Biological monitoring is surveillance conducted to ensure instream standards, or effluent permits, are being met using either the instream community or toxicity tests.

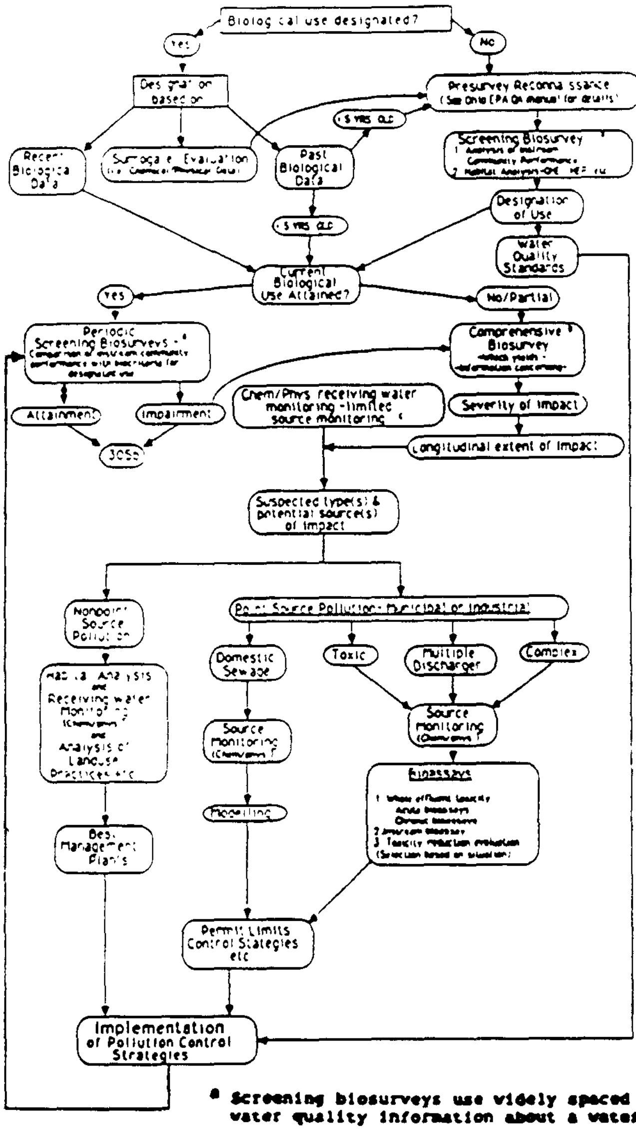
A successful biological assessment strategy to provide the information necessary to make correct regulatory decisions requires a knowledge and understanding of the strengths and weaknesses of each environmental tool employed. Table 1 depicts the ability of various chemical, physical, and biological assessment techniques to measure the key factors affecting the biological integrity of the surface waters. The relative use of each of these techniques, and their relative costs, should be carefully evaluated prior to any field sampling effort.

Bioassessments should be incorporated by the USEPA and State regulatory agencies into an integrated monitoring strategy that can accurately assess instream biological integrity and measure the success of pollution abatement programs. One method for utilizing biosurveys is presented in Figure 1, which is a conceptual framework implemented by one of the State regulatory agencies. The most important point of this framework is that when decisions are made on aquatic life use attainability and attainment, those decisions should be largely based on a direct determination of the aquatic community structure and function.

A technical support or guidance document should be developed by USEPA to identify how the biological criteria are implemented into the variety of regulatory and nonregulatory programs.

The data evaluation techniques currently employed in State programs should be the basis for establishing minimum data evaluation methods for new programs.

Education of State and Federal pollution control officials should be focused on the advantages and



a Screening biosurveys use widely spaced sampling sites to obtain general water quality information about a watershed.

b Comprehensive biosurveys require numerous sampling sites to document severity and extent of impacts. Sampling occurs at reference sites, sites up and downstream from point sources, and areas of expected changes in water quality.

c Biological communities react predictably to various water quality problems (e.g., toxicity, D.O. sags, siltation). As a result, biosurveys indicate the type of impact, and when coupled with receiving water monitoring provide greater resolution of the source(s) and type(s) of impacts

Fig. 1. A conceptual framework for assessing ambient biological performance and the success of implemented pollution control strategies (Ohio EPA 1987).

uses of biosurveys in the water programs. Education of regulatory agency biologists on the appropriate uses of biosurveys should be supported by USEPA Headquarters through Office of Research and Development (ORD) training. Education of the public and awareness of the biological ecosystem components are needed to urge legislators to initiate corrective programs through a citizens monitoring network.

The cost effectiveness of biosurveys for assessments has been demonstrated by the State of Ohio. Other States and regulatory agencies need to corroborate this finding to gain a more widespread acceptance of using biosurveys.

The use of automated databases and computer programs should be encouraged through Federal negotiations with States on their monitoring programs. The use of BIOS for storage and retrieval is highly recommended, and funding should be made available to support PC-based data manipulation software which would permit a free exchange of data between the States and the USEPA. Resources should also be allocated for a rigorous evaluation of the ERAPT program (Dawson and Hellenthal 1985).

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OVERVIEW OF STREAM QUALITY ASSESSMENTS AND STREAM CLASSIFICATION IN ILLINOIS

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Abstract

Since its creation in 1970, the Illinois Environmental Protection Agency (IEPA) has relied on aquatic macroinvertebrates in biosurveys conducted to evaluate degradation from point source dischargers. Early stream surveys utilized macroinveretebrates primarily as biological water quality indicators with data interpretation and pollution assessments made on the basis of presence or absence of intolerant organisms. Current water quality assessments are made using biotic community structure, tolerance ratings assigned to invertebrate taxa on a 0 to 11 scale, and Macroinvertebrate Biotic Index (MBI) values calculated from the equation: $MBI = (n_i * t_i)/N$. More recently, biosurveys have employed fish communities as biotic tools for stream quality evaluations, use support assessments mandated by the Clean Water Act, and for a cooperative interagency Biological Stream Characterization (BSC) process. The Index of Biotic Integrity (IBI) and associated fish community metrics are the foundation of data interpretation. IBI values were calculated by a program written in BASIC for the IBM-PC. Stream habitat quality assessments are now conducted in conjunction with fish monitoring utilizing a procedure which measures depth, velocity, and substrate type at eleven equally-spaced transects. Based on an equation derived from a multiple regression of IBI values and stream habitat data, the biotic potential of streams is estimated in the form of a predicted IBI value.

Introduction

The assessment of water quality by agencies responsible for pollution abatement has historically been the domain of the engineer, chemist, and microbiologist. Early efforts relied on analysis of dissolved oxygen, biological oxygen demand, pH, suspended solids, and fecal coliform bacteria. Use of chemical analysis was well suited for the evaluation of surface water quality and compliance of point source dischargers with numerical standards as these criteria were generally chemical in nature. Pollution assessment by chemical means, however, relies on collection of representative samples from a medium known to display frequent spatiotemporal variability. Chemical samples additionally provide no information regarding the degree to which abiotic factors influence biotic community structure and function.

Rationale

Passage of the Federal Water Pollution Control Act in 1972 (PL 92-500), and most recently, the Clean Water Act (CWA) Amendments of 1987, stressed assessment of not only water chemistry, but biotic integrity of the nation's waters. Focus on assessment of biotic integrity as a means of evaluating success of pollution control programs prompted the U.S. Environmental Protection Agency (USEPA) to issue guidelines for incorporation of biotic and abiotic factors into water body assessments for water quality standards

Table 1. Metrics used to assess fish communities in Illinois streams (from Karr et al. 1986).

Category	Metric	Scoring criteria		
		5	3	1
Species richness and composition	1. Total number of fish species	Expectations for metrics 1-5 vary with stream size and region and are discussed in the text.		
	2. Number and identity of darter species			
	3. Number and identity of sunfish species			
	4. Number and identity of sucker species			
	5. Number and identity of intolerant species			
	6. Proportion of individuals as green sunfish	<5%	5-20%	>20%
Trophic composition	7. Proportion of individuals as omnivores	<20%	20-45%	>45%
	8. Proportion of individuals as insectivorous cyprinids	>45%	45-20%	<20%
	9. Proportion of individuals as piscivores (top carnivores)	>5%	5-1%	<1%
Fish abundance and condition	10. Number of individuals in sample	Expectations for metric 10 vary with stream size and other factors and are discussed in the text.		
	11. Proportion of individuals as hybrids	0%	>0-1%	>1%
	12. Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies	0-2%	>2-5%	>5%

evaluation (USEPA 1982) and use attainment analyses (USEPA 1983).

To accomplish mandates of the Clean Water Act, the Illinois Environmental Protection Agency (IEPA) has conducted stream quality surveys since its creation in 1970. Surveys conducted in the early seventies relied solely upon aquatic macroinvertebrates as biological water quality indicators. Since the mid-seventies, stream surveys have also included water and sediment chemistry, and in recent years, the Agency has assessed fish communities and stream habitat in small wadeable streams. This paper summarizes the current use of aquatic macroinvertebrates and fish in IEPA stream quality assessments and the Biological Stream Characterization (BSC) process, and describes the development of a habitat assessment procedure for prediction of biotic potential in lotic environments.

Macroinvertebrates

Aquatic macroinvertebrates as defined by Weber (1973) are invertebrates large enough to be seen by the unaided eye, can be retained by a U.S. Standard No. 30 sieve (0.595 mm), and live at least part of their life cycles within or upon available aquatic substrates. Invertebrates included in this group typically consist of annelids, macrocrustaceans, aquatic insects, and mollusks (Isom 1978). Although macroinvertebrates were not routinely used in freshwater bioassays in the past (Weber 1973), they have been extremely useful in water quality monitoring through studies of community diversity and as indicator organisms (Resh and Unzicker 1975). Some of the advantages of using macroinvertebrates for environmental impact assessments include: limited mobility; relatively long life cycles; important members of aquatic food chains; sensitivity to a wide range of contaminants; known environmental requirements for key indicator groups; ubiquitous in distribution (occur where fish may not be present); and ease of collection.

While widely used for delineation of impacts caused by putrescible wastes, macroinvertebrates have also been used as indicators of heavy-metal pollution (Winner et al 1980), bioaccumulation (Mauck and Olsen 1977), and acidification (Mills and Schinder 1986).

Use of Macroinvertebrates in Illinois -- In 1970, the Illinois Environmental Protection Agency adopted and expanded a list of indicator organisms developed by Shiffman (1953) and continued use of a classification system in which streams were classified according to the percentage of intolerant organisms present. Using this procedure, the composition of a macroinvertebrate community at balanced stations consisted of more than 50% intolerant organisms; at unbalanced sites, less than 50% but more than 10% intolerant; at semipolluted sites, less than 10% intolerant; and community structure at polluted stations consisted of 100% tolerant organisms (Tucker 1961). The merits of this system for stream quality classifications were examined by Schaeffer et al. (1985).

Collection and Identification -- In 1982 IEPA biological staff made significant revisions to the IEPA Macroinvertebrate Tolerance List, updated field collections techniques, and adopted new data interpretation procedures for wastewater impact assessments (IEPA 1987). Qualitative collections of macroinvertebrates are made in the field using a No. 30-mesh sieve, D-frame net, and/or by hand picking of available substrates. Following collection of a sample, macroinvertebrate specimens are identified to a level consistent with survey objectives. In screening level surveys conducted to document impacts from wastewater facilities, invertebrates are identified in the field to family level. In selected special surveys or cooperative basin surveys conducted with the Illinois Department of Conservation (IDOC), macroinvertebrates are identified to the taxon and/or taxonomic level which has been assigned a tolerance rating by Agency biologists (Appendix Table A).

Data Handling -- Macroinvertebrate data are presently interpreted by an examination of community attributes: community structure, taxa richness, and use of the Macroinvertebrate Biotic Index (MBI). This index is a modification of a biotic index developed in Wisconsin (Hilsenhoff 1977, 1982). The MBI, similar to the Wisconsin index, provides a summation or average of tolerance values assigned to each taxon collected and weighted by abundance; low values indicate good water quality and high values degraded water quality. This index is on a 0 to 11 scale rather than the 0 to 5 scale originally proposed by Hilsenhoff. IEPA has also assigned tolerance ratings to several invertebrate groups not rated by Hilsenhoff: Turbellaria, Annelida, Decapoda, and the Mollusca. The Macroinvertebrate

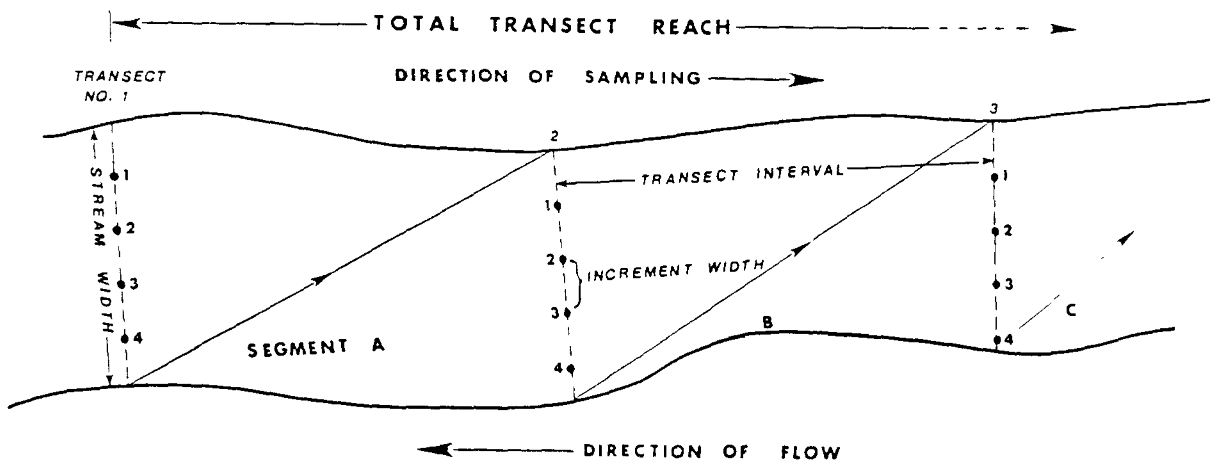

Fig. 1. Schematic diagram of IEPA habitat quality assessment procedure for wadeable streams. Sampling is initiated at the right edge of the water (REW) at transect 1. Depth, velocity and substrate measurements start at the proper increment width from REW (point 1) and sampling proceeds across transect. Additional transects are sampled at 10 yard intervals moving upstream (IEPA 1987).

Biotic Index is calculated by the following equation:

$$MBI = \sum (ni\ ti)/N$$

where: ni = No. individuals in each taxon
ti = Tolerance value for taxon
N = Total no. individuals

Fish

Over 180 species of fish have been recorded in Illinois (Smith 1979) and a majority of these species inhabit lotic environments. They occupy upper levels of aquatic food chains and are directly and indirectly affected by chemical and physical changes in their environment. While use of aquatic macroinvertebrates and water chemistry are integral components in the assessment of water quality and documentation of constituents causing impairment, the condition of the fishery is the most meaningful index of stream quality to the general public (Weber 1973).

Use of fish to assess biotic integrity of water resources has received increased emphasis in recent years by a number of investigators (Karr 1981; Hocutt 1981; Stauffer et al. 1976; Karr et al. 1986). Karr (1981) listed several advantages for using fish as indicator organisms in monitoring programs: life-history information is extensive for most species; fish communities generally include a range of species that represent a variety of trophic levels; fish are relatively easy to identify; both acute toxicity and stress effects can be evaluated; fish are typically present, even in the smallest streams and in all but the most polluted waters; and results of fish studies can be directly related to the fishable waters mandate of Congress.

IEPA Use of Fishery Data -- Early fish sampling efforts organized by IEPA were conducted largely to assess contaminant levels in selected fish populations in conjunction with biosurveys of Illinois river basins (Hite and King 1977). The Agency has subsequently placed greater emphasis on fish communities as indicators of stream quality. Starting in 1981, IEPA utilized fish data obtained in cooperative basin surveys by the Department of Conservation for water quality standards development, aquatic life use support assessments, stream classification, and to develop a stream habitat evaluation procedure. In 1986, the Agency initiated fish collections for the first time in an assessment of biotic integrity downstream from a large refinery complex in eastcentral Illinois (Hite et al. 1988).

Field Collection -- For stream quality assessments IEPA biologists typically collect fish with a combination of electrofishing and seining. Small wadeable streams are sampled using backpak or electroseine apparatus for 15 to 30 minutes. If additional sampling is required to obtain a representative sample, the length of sampling time is recorded for determination of catch per unit of effort. Three supplemental seine hauls with a 3/16 inch mesh seine are utilized at each site when suitable habitat exists. Larger streams are sampled using boat electrofishing gear for 30 minute periods (IEPA 1987). All fish collected are sorted, identified to species, and counted at the site. Those specimens which cannot be identified (eg., various cyprinids) are preserved in a 10 percent formalin solution for subsequent laboratory identification.

Data Interpretation -- Fisheries data are interpreted with the Index of Biotic Integrity and use of the 12 IBI metrics (Table 1; Karr et al. 1986). When fishery data does not allow calculation of a "pure" IBI value using all 12 metrics, an alternate Index of Biotic Integrity (AIBI) is calculated. Applicable metrics (e.g., number of species, intolerant individuals, etc.) of both the IBI and AIBI have been modified geographically for Illinois streams and expectations are determined by major river basin (Bertrand 1985). To expedite IBI calculations and fishery assessments made by biologists, the Agency developed a computer program for use on the IBM-PC (Kelly 1986). This program, updated in 1988 (Bickers et al. 1988), provides station location information, a summary of IBI metrics, the IBI or AIBI value (as appropriate), and a list of species collected (see Appendix Table B).

Stream Habitat Assessment

Biotic-Abiotic Relationship -- The abundance and distribution of individual species in lotic ecosystems is largely governed by geographically related physicochemical variables. Although aquatic life is found almost everywhere there is permanent water, each species has its own distribution or range; within that range, a species has unique environmental requirements and occurs in certain settings that are its habitat (Pflieger 1975).

Stream habitat consists of chemical and physical components. Both suitable water quality and desirable physical habitat (e.g., adequate depth, velocity, bottom substrate and cover) must exist to meet specific individual requirements. Both habitat components, while largely determined by geography, climate and local relief, may also be influenced by activities of man. In Illinois, few if any streams exist that have not been altered to some degree chemically or physically. These hydrological modifications, which include channelization and alteration of flow regimes, typically reduce the quality and quantity of habitat available for aquatic life, and ultimately biotic integrity. Indeed, physical alterations in the form of channelization have been reported to affect over 3400 stream miles in Illinois (Conlin 1976). In stream segments impaired by hydrological modifications, pollution control efforts to maintain and restore biotic integrity through water quality improvements may have limited success.

Instream physical habitat information was routinely recorded for all IEPA stream quality surveys in the past, but this limited data was subjective in nature. Because a systematic methodology for habitat analysis was not used in early IEPA stream surveys, it was often difficult to determine which habitat component - chemical or physical - was most limiting to aquatic communities.

Habitat Diversity

In 1982 a detailed stream habitat assessment procedure was adopted to complement fish, macroinvertebrate, water and sediment chemistry data normally collected in cooperative IEPA/IDOC basin surveys (Hite 1982). This method was predicated on the relation of habitat diversity (HD) to fish species diversity (FSD) demonstrated in several Midwest and Panama streams (Gorman and Karr 1978). This procedure was initially used in basin surveys conducted in the lower Kaskaskia, Sangamon, and Fox River Basins in 1982.

Habitat Diversity Field Methodology -- Stream habitat was measured in wadable streams along three dimensions considered important to fish. This methodology employed placement of transects along a study area with depth, velocity and substrate measured at equally spaced intervals on each transect. Location and length of the habitat study reach was identical in most cases to the IDOC fish sampling reach in cooperative basin studies. For 100 yard stream segments sampled by rotenone, transects were placed at 10 yard intervals starting from the upstream end of the study area; when available, the first transect was placed across a riffle area. This method resulted in placement of 11 equally spaced transects within the study area. Depth, velocity and substrate were recorded at equally spaced increments across each transect with increment spacing determined by mean stream width (Table 2). Water depth was measured with a USGS top-setting wading rod or fiberglass level rod to the nearest tenth of a foot (0.1 ft). Mean velocity at each transect increment was measured to 0.01 feet per second (ft/sec) with a Price AA current or pygmy meter at 0.6 total depth. Substrate or bottom type was recorded at each transect increment using appropriate substrate or bottom type codes. Habitat sampling

Table 2. Increment spacing as determined by mean stream width.

STREAM WIDTH (ft)	INCREMENT SPACING
$\bar{X}$ W $\leq$ 10	1
$\bar{X}$ W > 10 but $\leq$ 30	2
$\bar{X}$ W > 30 but $\leq$ 60	3
$\bar{X}$ W > 60 but $\leq$ 100	5
$\bar{X}$ W > 100	10

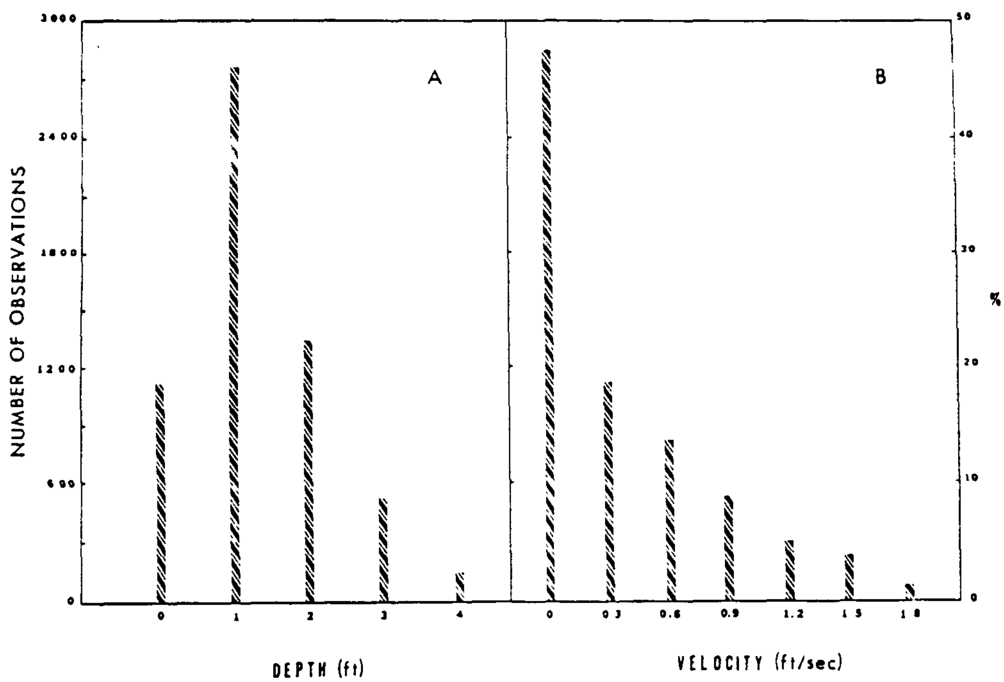

Fig. 2. Distribution of depth (A) and velocity (B) measurements at 52 lower Kaskaskia River basin sites, summer 1982.

Table 3. Instream habitat categories developed from 52 lower Kaskaskia River sites, 1982.

CATEGORY	DEPTH (ft)	VELOCITY (ft/sec)	SUBSTRATE	(INCHES	- mm)
1	0 - 0.5	-0.15 - 0.15	SILT-MUD	<0.002	<0.062
2	0.5 - 1.5	0.15 - 0.45	SAND	0.02-0.08	0.62-2
3	1.5 - 2.5	0.45 - 0.75	GRAVEL	0.08-2.5	2-64
4	2.5 - 3.5	0.75 - 1.05	RUBBLE	2.5-9.8	64-250
5	3.5 - 4.5	1.05 - 1.35	BOULDER	>9.8	250-4000
6	>4.5	1.35 - 1.65	BEDROCK		
7		>1.65	CLAYPAN		
8			PLANT DETRITUS		
9			VEGETATION		
10			LOGS		

was initiated at the right edge of water (REW) at the most downstream transect (transect 1), and proceeded in an upstream direction until HD dimensions were recorded at each increment in the 11 transect reach (Fig. 1).

In summer 1982 habitat diversity measurements were recorded at over 5900 points at 52 lower Kaskaskia fish collection sites. Study areas varied from unmodified natural stream segments, to fairly recent or older channelized areas. Stream size ranged from a few small 2nd order streams to much larger 5th or 6th order streams. Over 2700 (47%) depth measurements were within a range of 0.5 to 1.5 feet (Fig. 2). Stream velocities ranged from over 2.0 ft/sec to no detectable flow -- a common occurrence in the lower Kaskaskia Basin. Approximately 70% of all velocity measurements were less than 0.5 ft/sec. By bottom type or substrate category, over 70% of all observations consisted of silt-mud, sand or gravel.

HD Data Analysis -- Using the mainframe and discriminant analysis program available in the Statistical Analysis System (SAS 1982) package at Southern Illinois University at Carbondale, depth and velocity data were analyzed to develop categories for calculation of habitat diversity. From this analysis, six depth and seven velocity categories were identified (Table 3). With the 11 substrate categories, a possibility of 462 combinations existed for calculation of HD using the Shannon-Weiner equation. Habitat diversity values for each lower Kaskaskia River site were plotted against FSD and IBI values

Table 4. Substrate, bottom type, and other metrics used in IEPA habitat assessment procedure (modified from IEPA 1987).

CODE	SUBSTRATE	PARTICLE SIZE	OTHER METRICS
1	Silt/mud	<0.062 mm	Depth (ft)
2	Sand	0.062 - 2 mm	
3.1	Fine gravel	2 - 8 mm (0.08 - 0.3 in)	Velocity (ft/sec)
3.2	Medium gravel	8 - 16 mm (0.3 - 0.6 in)	
3.3	Coarse gravel	16 - 64 mm (0.6 - 2.5 in)	Instream Cover (%)
4.1	Small cobble	64 - 128 mm (2.5 - 5 in)	
4.2	Medium cobble	128 - 256 mm (5 - 10 in)	Pool (%)
5	Boulder	256 - 4000 mm (>10 in)	
6	Bedrock	Solid Rock	Shading (%)
	BOTTOM TYPE		
7	Claypan - compacted soil		
8	Plant detritus		
9	Vegetation		
10	Submerged logs		
11	Other __________________		

Table 5. Habitat metrics used in stepwise multiple regression analysis.

	SUBSTRATE	
WATER QUALITY (WQI)		
DISCHARGE (CFS)	SILT-MUD	BEDROCK
MEAN DEPTH (ft)	SAND	CLAYPAN
MEAN VELOCITY (ft/sec)	GRAVEL	PLANT DETRITUS
POOL (%)	COBBLE	VEGETATION
INSTREAM COVER (%)	BOULDER	SUBMERGED LOGS
SHADING (%)		

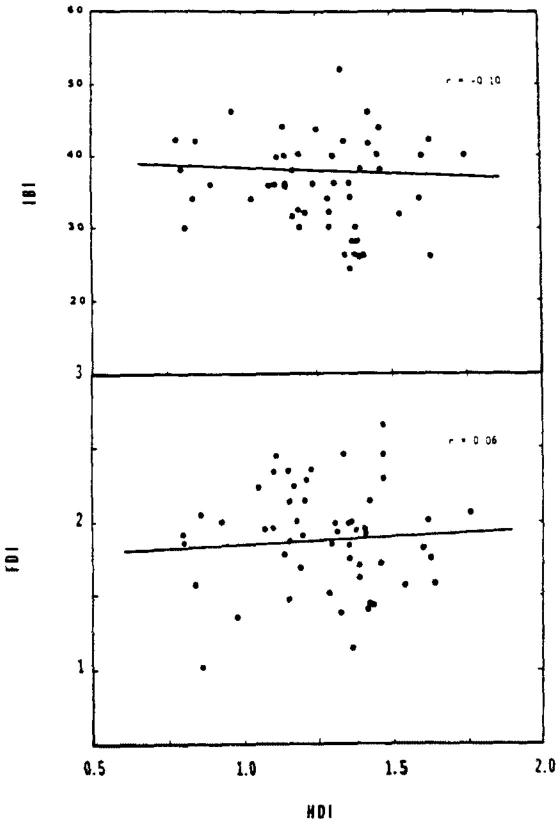

Fig. 3. Relationship of the Index of Biotic Integrity (A) and fish species (B) to habitat diversity at 52 lower Kaskaskia River basin sites, 1982.

calculated for the same location (Fig. 3). Simple linear regression analysis confirmed what wasvisually evident: no significant relationship existed between HD and FSD or IBI for the 52 lower Kaskaskia Basin sites. It was the authors' opinion that the removal of the few lower Kaskaskia Basin sites thought to be water quality limited would not have notably improved this relationship.

<u>Biotic Potential Assessment Strategy</u> -- Following evaluation of HD and the inability to demonstrate any statistical relationship between HD and FSD or IBI, alternative habitat assessment and data analysis techniques were examined. This strategy involved four basic steps: 1). development of a statewide data base consisting of sites with fishery, water quality, and habitat data collected within a similar time frame; 2). determination of sites where biotic communities were impacted or limited by water quality, eliminating these sites from the data base; 3). use of statistical analysis to determine which habitat variables were most important in determining biotic integrity as measured by IBI, and; 4). development of an equation which predicted IBI from habitat metrics.

<u>Field Methods</u> -- Habitat evaluation efforts in 1983 in the upper Kaskaskia River Basin and other basin surveys utilized similar field assessment methods employed for habitat diversity but placed less emphasis on velocity measurements -- the most time consuming aspect of habitat assessment. Several other habitat metrics, however, were added to habitat surveys conducted in 1983 and subsequent years: three substrate categories, estimates of instream cover, riffle-pool development, and shading (Table 4). In general, habitat assessments conducted in conjunction with cooperative interagency basin surveys were restricted to flowing, wadeable streams and sampling was conducted with streams at base or low flow condition.

<u>Data Base Development</u> -- To develop the data base necessary to determine habitat-biotic integrity relationships, water quality, habitat, and IBI values from about

250 sites in five Illinois river basins were entered into the mainframe at Southern Illinois University at Carbondale. In addition to the metrics determined from stream habitat evaluations, stream discharge and water quality information were added. Water quality was measured by a STORET-generated index designated as WQI; this index is on a 0 to 100 scale with higher values indicating more degraded water. Water quality parameters used with this index -- temperature, dissolved oxygen, pH, total phosphorus, turbidity, conductivity, and ammonia nitrogen -- were selected on the basis of a matrix analysis which correlated water quality constituents against macroinvertebrate biotic index values (Kelly and Hite 1984).

Statistical Analysis -- Fifteen stream habitat metrics, WQI, and discharge data (Table 5) were subjected to multiple regression analysis using the PROC STEPWISE procedure in SAS (1982). When all data were included in the analysis, water quality as measured by WQI, was the most important variable affecting biotic integrity. By selectively eliminating sites from the data set on the basis of WQI values, an equation was generated that selected habitat variables in preference to water quality. It was found when only sites with WQI values less than 60 were included in the regression analysis, habitat variables became more important in explaining variance in IBI values. Following elimination of all sites exhibiting WQI values $\geq$ 60, 149 sites from the five river basins remained in the data base with a large number of these sites from the Kaskaskia River Basin. Remaining sites were not considered to be water quality limited; any biotic integrity perturbations now evident were attributed to be a function of physical habitat quality.

Multiple regression analyses of habitat metrics and IBI values for the 149 sites indicated four metrics appeared to exert the greatest influence on biotic integrity as measured by IBI. In order of importance, habitat variables accounting for the greatest percent variance in IBI values included: 1). Percent silt-mud (r^2 = 0.163), 2). Percent claypan (r^2 = 0.216), 3). Mean stream width (r^2 = 0.252), 4). Percent pool (r^2 = 0.282).

For biotic integrity prediction it was necessary to develop either: 1). a matrix type evaluation procedure with which applicable habitat metrics influencing IBI would be assigned arbitrary weights for stream reach classification, or 2). use the regression equation derived from the SAS PROC STEPWISE procedure. To expedite use of habitat data for aquatic life use support assessment in the 1986 IEPA 305(b) report, the later course was selected. The regression analysis yielded the following equation for prediction of biotic potential as defined by a predicted IBI value:

Predicted IBI = 40.1-(0.126*siltmud)-(0.123*claypan)+(0.0424*pool)+(0.0916*width)

Using the biotic potential equation or PIBI, predicted values can range from about 27 to 53, or from a BSC rating of a Limited Aquatic Resource (D) to a Unique Aquatic Resource (A). When applied to typical Illinois stream habitat data from 3rd to 6th order streams, most PIBI values routinely fall between 35 and 50. For 102 sites sampled in the Kaskaskia River

Basin in 1982 and 1983, having a mean stream order of 4.3, the mean predicted IBI was 40.4 (Kelly et al. 1988).

Current Biosurvey Programs

Use of macroinvertebrate, fish, and habitat data in current IEPA surface water monitoring programs falls into three general categories: 1) stream quality surveys for documentation of impacts from point source dischargers; 2) basin surveys for determination of aquatic life use support attainment and Biological Stream Characterization; and 3) Special Surveys. Point Source-Related Surveys. The majority of IEPA biosurveys are conducted to document stream conditions in the vicinity of industrial and municipal wastewater dischargers. One such program, termed Facility-Related Stream Surveys (FRSS), consists of the collection of biotic, water chemistry, stream flow and habitat quality data upstream and incrementally downstream from municipal or industrial discharges. Macroinvertebrates are utilized to assess existing stream quality and/or document degradation from the discharge. Fish are occasionally collected for contaminant analyses or for stream classification purposes. Water chemistry parameters include water temperature, dissolved oxygen, biochemical oxygen demand (BOD), chemical oxygen demand (COD), un-ionized ammonia nitrogen, nitrate-nitrite nitrogen, and total phosphorus, and total metals. Biotic and chemical data generated from FRSS are used to assess: representativeness of Agency and discharger effluent monitoring data, stream quality impacts, the need for additional wastewater treatment, and appropriate NPDES permit limitations.

Biological Stream Characterization -- Historically, grant monies for construction or renovation of wastewater treatment facilities in Illinois have been allocated to metropolitan areas either willing to enter the grant process or able to fund their portion on construction costs. Prioritization for funding, while based on many factors, rarely had any relationship to potential aquatic life use or value of the aquatic resource to be protected. To accomplish Clean Water Act objectives and ensure that important aquatic resources are considered in the allocation of limited pollution control monies and staff resources, classification of Illinois streams was necessary.

In 1983, IEPA biologists proposed a stream classification system based on the type and condition of the fishery and macroinvertebrate community structure. This provisional classification methodology was provided to IDOC stream biologists for review and was subsequently applied to the Fox River Basin in northern Illinois in fall 1983. In Spring 1984, biologists from IEPA and IDOC met and formed the Biological Stream Characterization (BSC) Work Group to address biotic classification of Illinois streams.

The BSC Work Group developed a provisional five-tier stream classification system in 1984. This stream classification system is based largely on attributes of lotic fish communities using the Index of Biotic Integrity (Table 6). When suitable fishery data are not available for calculation of an IBI value, the site may be classified on the basis of the

Table 6. Biological Stream Characterization (BSC) criteria for the classification of Illinois streams.

RESOURCE DESCRIPTION:	UNIQUE AQUATIC RESOURCE	HIGHLY VALUED AQUATIC RESOURCE	MODERATE AQUATIC RESOURCE	LIMITED AQUATIC RESOURCE	RESTRICTED USE AQUATIC RESOURCE
BIOTIC METRIC / CLASS:	A	B	C	D	E
FISHERY					
Index of Biotic Integrity (IBI) or Alternate (AIBI)	51 - 60	41 - 50	31 - 40	21 - 30	< 20
Sport Fishery Value		Good fishery for walleye, sauger, smallmouth, spotted, or largmouth bass, northern pike, white bass, crappie, catfish, rock bass, or put and take trout fishery.	Smaller species of sport fish predominate in sport catch. Bullhead/sunfish, carp fishery. Diverse forage fish community may be present.	Carp or other less desirable species support fishery. Few if any fish of other species caught.	No sport fishery. Few fish of any species.
Spawning or Nursery Value		Tributary to an "A" stream, or used as nursery by above sport fish species.	Nursery or rearing area for common sport fish. Young of year or juveniles of above species common in fish samples.	Few if any young of year or juveniles of any sport species present.	No young of year or juveniles of sport species present.
MACROINVERTEBRATES					
Macroinvertebrate Biotic Index (MBI)	N/A	N/A	N/A	≥ 7.5 ≤ 10.0	> 10.0
Community Structure	N/A	N/A	N/A	Predominant macroinvertebrate taxa/individuals consist of facultative and/or moderate organisms. Intolerant organisms are sparse or may be absent.	Intolerant organisms absent; benthic community consists nearly all tolerant forms, or no aquat macroinvertebrates may be present.
Species Richness	N/A	N/A	N/A	Notably lower than expected for geographic area, stream size or available habitat; usually limited to a moderate or few	Restricted to few taxa, or no taxa present.

sport fishery value. Macroinvertebrates are factored into the BSC process when fishery data are not available and are used to assign a limited or restricted BSC rating (Class D or E respectively) to stream segments greater than five miles in length. When using macroinvertebrate data for stream classification purposes, biologists may utilize MBI values and/or other community metrics such as species richness or community composition.

Aquatic Life Use Support Assessment -- In addition to use in BSC, both fishery and macroinvertebrate data are used for aquatic life use support assessments required by Section 305(b) of the Clean Water Act. In accordance with federal guidance (USEPA 1987), use support assessments are completed for each stream reach sampled in conjunction with cooperative IEPA/IDOC intensive basin surveys. The degree to which Illinois streams support designated uses is determined using a combination of biotic and abiotic data, intensive survey field observations, and professional judgment. Because it is felt that aquatic life is the best indicator of the CWA goals of fishable and swimmable waters, the use support process focuses on biotic data and Biological Stream Characterization (BSC) ratings when available. Biotic data consist of fishery and macroinvertebrate community data which are evaluated using the index of biotic integrity (Karr et al. 1986) and the IEPA Macroinvertebrate Biotic Index (MBI), respectively. Abiotic data includes water chemistry, fish tissue analysis, sediment chemistry, and physical habitat metrics.

Four levels of use support assigned to Illinois streams include: Full, Partial/Minor, Partial/Moderate, and Nonsupport (IEPA 1988). A fifth category, Full/Threatened, is occasionally used to designate waters presently considered in full support but likely to change in the future because of changing land use patterns, new point sources, or a continued decline in water quality. Where fish, stream habitat and water quality data are available for the same site, the use support category is determined using a flow chart (Fig. 4). For waters with limited data available, assessments are made with general criteria provided in a use support classification matrix (Table 7). Because the 305(b) use support assessment process uses both fish and macroinvertebrate data, use support groups closely resemble BSC categories. The general relationship of the five BSC categories to use support assessment levels and other IEPA assessment metrics and criteria is depicted in Table 8.

Classification of Fishable Waters -- Section 305(b) of the Clean Water Act also requires assessment of the degree to which CWA fishable/swimmable goals have been attained. These goals are considered separate and independent criteria from designated use assessment guidelines (USEPA 1987). USEPA has defined fishable goals for the 305(b) process as "providing a level of water quality consistent with the goal of protection and propagation of a balanced population of shellfish, fish and wildlife." In Illinois, criteria for evaluating attainment of aquatic life use has incorporated selected biotic indices or classification systems:

Table 1. Criteria and use support classification matrix for Illinois streams.

BASIS OF ASSESSMENT	ASSESSMENT DESCRIPTION	LEVEL OF USE SUPPORT: FULLY SUPPORTING	PARTIAL/MINOR	PARTIAL/MODERATE	NONSUPPORTING
EVALUATED	No ambient or intensive data available. Assessment based on historic data, location, similarity of area to monitored waters within geographic area or ecoregion. Assessments are predictions which have not been verified by recent monitoring data.	No point or nonpoint sources are present that could interfere with use support. Physiographic similarities of area to monitored waters or general familiarity of water or reach indicates full support.	Some water quality criteria excursions thought to exist or minor impact to aquatic life use support predicted on basis of known point or nonpoint sources or physical habitat limitations.	Moderate aquatic life impairment predicted on basis of known point or nonpoint sources or physical habitat limitations.	Severe aquatic life impairments predicted on basis of known point sources.
MONITORED					
Biosurvey Data	Fish or Macroinvertebrate community assessed by professional biologist. Assessment protocol includes evaluation of species richness, community structure and/or biotic integrity evaluation, and a comparison of biotic quality with biotic potential as measured by habitat data.	No significant modification of aquatic community structure and function (<10%). Community within expectations for stream size and physiographic region or ecoregion. Index of biotic integrity (IBI) usually >41 or within 4 points of biotic potential (PIBI) predicted by stream habitat assessment. macroinvertebrate biotic index (MBI) values usually <6.0.	Some modification of aquatic community apparent, resulting 10-25% decline in species richness, intolerant forms, number of individuals or applicable biotic index values; similar increase in number of non-sensitive forms may be evident. IBI values generally range from 31-40 or may be slightly lower than PIBI (>4 but ≤8). MBI values generally range from 6.0-7.5.	Notable deterioration of aquatic community evident; 25-50% decline in biotic community quality metrics and/or commensurate increase in number of nonsensitive individuals. IBI values typically <30, or notably lower than biotic potential (PIBI - IBI > 8 but < 14). MBI values generally range from 7.5-10.0.	Severe deterioration of aquatic community; >50% reduction in species richness, intolerant forms, number of individuals and/or applicable indices of biotic integrity. A similar increase in number of tolerant forms may be evident. Aquatic life may not be present. IBI values generally ≤23 or >14 points lower than PIBI predicted from habitat. MBI values usually >10.0.
Water Chemistry	Fixed station ambient or intensive basin water quality sampling. Assessment based on water quality index values, evaluation of raw water chemistry data and/or water quality criteria excursions. Used when biotic data are not available and/or to supplement bio-survey data.	WQI values generally <30; index values influenced primarily by phosphorus, total suspended solids (TSS) or minor DO excursions. TSS concentrations usually <25 mg/l. Pesticides or priority pollutants usually not present or detected only at trace levels. Other constituents usually within State standards.	WQI values typicaly range from from 30-50. Index values driven primarily by nutrients/TSS, or minor DO, pH excursions. TSS generally range from 25-80 mg/l. Pesticides or priority pollutants may be may be present but at low levels. Some State WQ standards may occasionally be exceeded.	WQI values usually range from 50-70. Index values may be influenced by several constituents including DO, pH, or other parameter groups. TSS generally exceed 80 mg/l. Pesticides or priority pollutants not at levels of concern when present. State WQ standards for selected constituents frequently exceeded.	WQI values generally >70; in addition to DO or pH, index values are usually influenced by several parameter groups including metals or inorganic toxicity. Extreme TSS levels (>400 mg/l) may occur. Pesticides or priority pollutants may be found at levels of concern. Water quality standards/criteria for critical aquatic life metrics routinely exceeded.
Fish Tissue	Cooperative interagency fish contaminant monitoring program; samples collected from fixed statewide network and/or from intensive studies. Tissue analysis conducted for human health implications and contaminant trend monitoring.	Organochlorine compounds in fish tissue not detected or occasionally present at trace concentrations.	Organochlorine compounds routinely detected in fish populations but contaminants not found at levels of concern.	Moderate concentrations of priority organochlorine compounds routinely detected in fish community with some species and size classes occasionally exceeding USFDA tolerance levels. Advisories for limited consumption of selected fish or sizes may be issued.	Concentrations of priority organochlorine compounds consistently found in fish community at or higher than USFDA tolerance levels. 'Consumption' advisories issued.

Table 6. SUMMARY OF USE SUPPORT ASSESSMENT CRITERIA FOR ILLINOIS STREAMS

USE SUPPORT DESCRIPTION / CRITERIA

	USEPA	FULL SUPPORT		PARTIAL SUPPORT MINOR	PARTIAL SUPPORT MODERATE	NON-SUPPORT
STREAM QUALITY CATEGORY	GENERAL STREAM/WATER QUALITY CONDITION	Excellent	Very Good	Fair-Good	Poor	Very Poor
	IEPA/IDOC BIOLOGICAL STREAM CHARACTERIZATION (BSC)	Unique Aquatic Resource	Highly Valued Resource	Moderate Aquatic Resource	Limited Aquatic Resource	Restricted Aquatic Resource
ASSESSMENT METRIC/INDEX	FISH/Index of Biotic Integrity (IBI/AIBI)	51-60	41-50	31-40	21-30	< 20
	BENTHOS/Macroinvertebrate Biotic Index (MBI)	< 5.0	5.0-6 0	6.0-7 5	7.5-10.0	> 10.0
	WATER CHEMISTRY/ STORET Water Quality Index (WQI)	0-10	10-30	30-50	50-70	> 70
	WATER CHEMISTRY/ Total Suspended Solids (TSS/mg/l)	< 10	10-25	25-80	80-400	> 400
	STREAM HABITAT/ Potential Index of Biotic Integrity (PIBI)	51-60	41-50	31-40	< 31	
	STREAM SEDIMENT/ IEPA Stream Sediment Classification	Nonelevated	Nonelevated -Slightly Elevated	Slightly Elevated	Elevated -Highly Elevated	Extreme

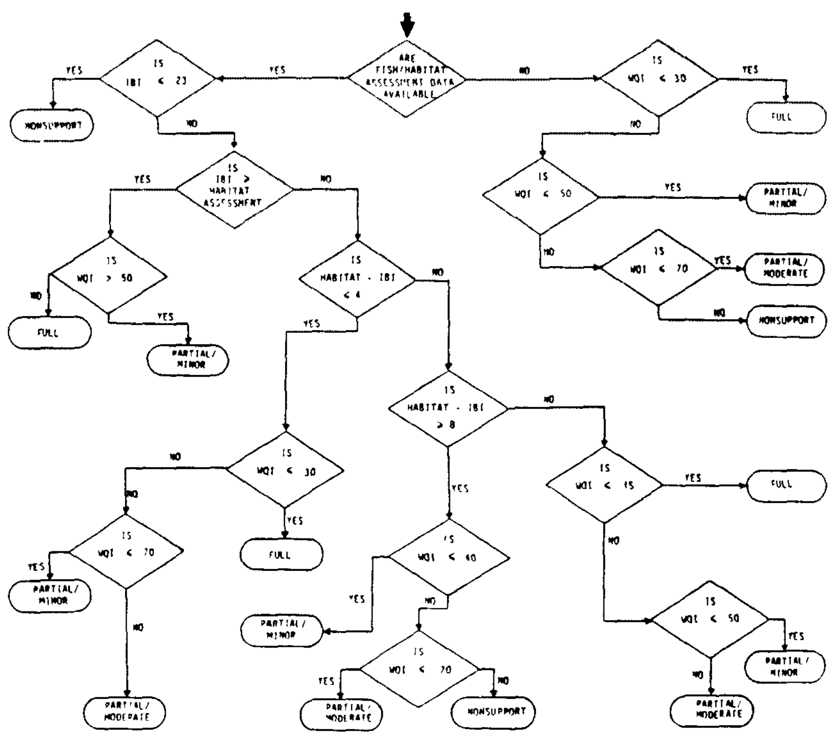

Fig. 4. Aquatic life use support assessment flow chart for fish, habitat, and water quality data.

IBI, MBI, and Biological Stream Characterization. For assessment using macroinvertebrate data, streams are considered not meeting fishable goals if MBI values are $\geq$ 10.

Special Surveys -- Special stream surveys are routinely conducted in response to catastrophic events (e.g., spills of toxic materials), mineral extraction, nonpoint source problems including agriculture and abandoned mines, and in support of enforcement proceedings. The scope and design of the survey is dependent upon the nature of the stream system and type of contaminant. In addition to use of biotic and chemistry, special surveys may utilize sediment chemistry analysis as a mechanism for screening organochlorine compound and heavy metal contaminants. The extent of stream sediment contamination is determined from an Agency sediment chemistry classification (Kelly and Hite 1984).

General Biosurvey Problem Areas

Unfortunately in many states, biosurveys and data are not utilized to the extent they should be in pollution control programs. Biosurvey problems which are evident from discussions with biologists involved with water pollution programs nationwide include: compliance and facility inspections programs measure surrogates of aquatic resource quality (i.e. effluent quality) instead of the resource itself; biotic data are frequently not considered in decisions regarding permitting activities, scheduling facility inspections, and in the awarding of construction grants; biotic data are rarely used in the decision process when siting new wastewater treatment facilities or in the relocation of existing point source discharges; bioassays appear to have been overemphasized by USEPA in recent years as the biomonitoring tool of choice. Bioassays like other types of biomonitoring, have their place in pollution control programs, but like effluent monitoring, do not measure stream biotic integrity and are useful only when representative samples are taken.

The emphasis on use of effluent data in lieu of actual stream quality data by facility inspection, compliance assurance and permitting programs occurs because: 1) the lack of biocriteria has resulted in federal mandates that require states to measure success of pollution control programs at the end of the pipe rather than by improvements in stream biotic integrity; 2) stream biocriteria have not been developed because of reliance on existing effluent and water quality standards; 3) water pollution control programs are usually managed by individuals who are technology and hardware orientated and who thus focus on facilities; 4) pollution control administrators and engineers frequently do not understand biological data; and 5) Many state water pollution control programs do not have legislative authority to address problems that are nonpoint source in nature or to manage water resources logically -- on a watershed basis.

Discussion

The macroinvertebrate, fish, and stream habitat assessment procedures presented here represent the current use of these assessment tools in biosurveys conducted by the Illinois Environmental

Protection Agency. The assessment of biotic and abiotic factors as a means of conducting lotic resource inventories, pollution control appraisals, or deriving aquatic classifications is an evolutionary process. Assessment procedures will change just as certainly as will the technology and advances in the aquatic sciences which will necessitate this change. Modification of assessment techniques will also be necessary to accommodate institutional change at the state level and never-ending change from the federal perspective.

Aquatic Biota as Environmental Indicators

Fish Community Evaluations -- Assessment of biotic integrity using fish populations will undoubtedly receive more emphasis in IEPA biosurveys in ensuing years. Fish populations integrate both chemical and physical perturbations which affect stream quality and are ideal environmental indicators from the general public perspective. Many additional advantages exist for use of this group as biological indicators (Karr, et al. 1986; Hocutt 1981). Their use in existing programs such as contaminant assessments, aquatic life use support determinations, Biological Stream Characterization, and probable use for future nonpoint source assessments assure a prominent role in Agency monitoring activities.

Benthic Macroinvertebrates -- Macroinvertebrates will continue to be the primary biotic tool used for IEPA point source related impact assessments. The advantages of using these indicator organisms to assess differences in stream quality in an "upstream-downstream" fashion and to demonstrate effectiveness of pollution control programs via pre- and post-wastewater facility construction surveys are well established (Cairns et al. 1972). The macroinvertebrate biotic index currently employed provides an adequate impairment assessment when applied to data collected from streams which receive organic wastes discharged from the typical municipal wastewater treatment facility. Utility of this index diminishes, however, when applied to data collected from streams impaired by inorganic suspended solids, toxic contaminants, and/or other abiotic perturbations. In the future, development or adoption of a multi-parameter macroinvertebrate index conceptually similar to IBI is desirable. The advantages of incorporating several attributes of community well being (e.g., taxa richness, trophic composition, etc.) into one index are obvious as interpretation of biological data must frequently be condensed down to simplistic terms or index values for understanding by water resource managers with little time or expertise to delve into complex biological data. Development and use of such an index has been initiated by the Ohio EPA (Rankin 1986) and recently advocated by USEPA as a rapid bioassessment protocol (Plafkin et al. 1987).

Stream Habitat Assessment and Classification

Habitat Diversity -- The concept that fish species diversity is ultimately related to structural and hydrological complexity in lotic ecosystems is widely accepted among practicing aquatic biologists. Failure of the habitat diversity procedure to demonstrate any significant relationship to either IBI or FSD may have been more attributable to a data set restricted geographically than error in theory. The relative homogeneity (i.e., lack of diversity) of stream habitat in the largely agricultural lower Kaskaskia River Basin may have been in part responsible for lack of any relationship. A true test of this relationship would be better assessed by a much larger data base of widely differing fish communities and habitat types -- something that was lacking in the data set used.

Habitat Assessment -- Evaluation of stream habitat quality will continue to be an integral component of IEPA biosurveys. Prediction of biotic potential (PIBI) by use of a single multiple regression equation is not without problems. The present predictive equation was essentially based on the relationship of the Index of Biotic Integrity to physical habitat variables in third to fifth order central Illinois streams -- primarily the Kaskaskia River Basin. This central Illinois region, designated as the Central Corn Belt Plains Ecoregion (Omernik 1987), encompasses about 75% of the State; because of general physiographic similarities in this ecoregion, use of the PIBI equation may be applicable to many smaller streams in this area. Caution is suggested, however, in use of this equation for predicting IBI in physiographically dissimilar regions in Illinois, or elsewhere. Ultimately, continued use of a predictive equation for use support assessment or Biological Stream Characterization will require a specific equation be developed for each ecoregion, physiographic region, or river basin in Illinois where this procedure is to be used.

An array of habitat evaluation and data interpretation techniques have been developed for both warmwater and coldwater lotic systems, although certainly emphasis has been placed on the latter. The habitat and data assessment procedures detailed here are by no means considered state of the art or the best assessment techniques. The need for uniform habitat assessment techniques for similar geographic areas and program objectives is evident and has prompted formation of stream habitat assessment standardization committees by the American Fisheries Society at the national and more recently division level. These efforts have been proceeded by publication of proceedings from at least two major habitat evaluation-related symposia (Armantrout 1981; USF&W 1977) and by significant efforts at habitat quantification for aquatic life instream flow needs (Bovee 1982). Elaborate procedures for documentation of habitat requirements at the species level (USF&W 1980) have also been developed as have excellent and detailed methods for the assessment of stream habitat metrics (Platts et al. 1983).

Biological Stream Characterization

Historically, classification of streams in this country has been based on a multitude of biotic and abiotic variables. In Illinois, fish community characteristics have been used to rate the quality of major Illinois river basins (Smith 1971). Recent use of the Biological Stream Characterization (BSC) system by the Illinois

Environmental Protection Agency and Department of Conservation has emphasized biotic integrity measured by IBI and/or the value of the sport fishery resource. BSC is intended to serve the somewhat different objectives of two state agencies: one delegated authority for regulation of water quality (IEPA) and the other, management of aquatic life in Illinois (IDOC). It is therefore not surprising that BSC does not totally address the needs of either agency as well as a classification system dedicated to a single agency's specific needs. Because many important sport fishes in Illinois -- notably the centrarchids and ictalurids -- are generally considered fairly tolerant fishes, BSC ratings predicated on sport fishery values may not accurately reflect ambient water quality or habitat quality. In the future it will be necessary to evaluate present IBI numerical ranges used for BSC ratings and their relationship to water quality; and finally, it may be necessary to incorporate other stream quality metrics into BSC such as water and habitat quality before the full potential of this classification procedure is realized and ultimately used by Illinois water resource managers.

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Appendix A. ILLINOIS ENVIRONMENTAL PROTECTION AGENCY
MACROINVERTEBRATE TOLERANCE LIST

MACROINVERTEBRATE	TOLERANCE VALUE
PLATYHELMINTHES	
TURBELLARIA	6
ANNELIDA	
OLIGOCHAETA	10
HIRUDINEA	8
Rhynchobdellida	
Glossiphoniidae	8
Piscicolidae	7
Gnathobdellida	
Hirudinidae	7
Pharyngobdellida	
Erpobdellidae	8
ARTHROPODA	
CRUSTACEA	
ISOPODA	
Asellidae	6
Caecidotea	6
brevicauda	6
intermedia	6
Lirceus	4
AMPHIPODA	
Hyalellidae	
Hyalella	
azteca	5
Gammaridae	
Bactrurus	1
Crangonyx	4
Gammarus	3
DECAPODA	
Cambaridae	5
Palaemonidae	
Palaemonetes	4
INSECTA	
EPHEMEROPTERA	
Siphlonuridae	
Ameletus	0
Siphlonurus	2
Oligoneuriidae	
Isonychia	3
Metretopodidae	
Siphloplecton	2
Baetidae	
Baetis	4
brunneicolor	4
flavistriga	4
frondalis	4
intercalaris	7
longipalpus	6
macdunnoughi	4
propinquus	4
pygmaeus	4
tricaudatus	1
Callibaetis	4
fluctuans	4
Centroptilum	2
Cloeon	3
Pseudocloeon	4
dubium	4
parvulum	4
punctiventris	4
Heptageniidae	
Arthroplea	3
Epeorus	1
vitreus	0
Heptagenia	3
diabasia	4
flavescens	2
hebe	3
lucidipennis	3
maculipennis	3
marginalis	1
perfida	1
pulla	0
Rhithrogena	0
Stenacron	4
candidum	1
gildersleevei	1
interpunctatum	4
minnetonka	4
Stenonema	4
annexum	4
ares	3
exiguum	5
femoratum	7
integrum	4
luteum	1
mediopunctatum	2
modestum	3
nepotellum	5
pudicum	2
pulchellum	3
quinquespinum	5
rubromaculatum	2
scitulum	1
terminatum	4
vicarium	3
Ephemerellidae	
Attenella	2
Danella	2
Drunella	1
Ephemerella	2
Eurylophella	4
Seratella	1
Tricorythidae	
Tricorythodes	5
Caenidae	
Brachycercus	3
Caenis	6
Baetiscidae	
Baetisca	3
Leptophlebiidae	
Choroterpes	2
Habrophlebiodes	2
americana	2
Leptophlebia	3
Paraleptophlebia	2
Potamanthidae	
Potamanthus	4
Ephemeridae	
Ephemera	3
simulans	3

MACROINVERTEBRATE	TOLERANCE VALUE
Hexagenia	6
limbata	5
munda	7
Palingeniidae	
Pentagenia	4
vittigera	4
Polymitarcyidae	
Ephoron	2
Tortopus	4
ODONATA	
ANISOPTERA	
Cordulegasteridae	
Cordulegaster	2
Gomphidae	
Dromogomphus	4
Gomphus	7
Hagenius	3
Lanthus	6
Ophiogomphus	2
Progomphus	5
Aeshnidae	
Aeshna	4
Anax	5
Basiaeschna	2
Boyeria	3
Epiaeschna	1
Nasiaeschna	2
Macromiidae	
Didymops	4
Macromia	3
Corduliidae	
Cordulia	2
Epitheca	4
Helocordulia	2
Neurocordulia	3
Somatochlora	1
Libellulidae	
Celithemis	2
Erythemis	5
Erythrodiplax	5
Libellula	8
Pachydiplax	8
Pantala	7
Perithemis	4
Plathemis	3
Sympetrum	4
Tramea	4
ZYGOPTERA	
Calopterygidae	
Calopteryx	4
Hetaerina	3
Lestidae	
Archilestes	1
Lestes	6
Coenagrionidae	
Amphiagrion	5
Argia	5
moesta	5
tibialis	5
Enallagma	6
signatum	6
Ischnura	6
Nehalennia	7

MACROINVERTEBRATE	TOLERANCE VALUE
PLECOPTERA	
Pteronarcyidae	
Pteronarcys	2
Taeniopterygidae	
Taeniopteryx	2
Nemouridae	
Nemoura	1
Leuctridae	
Leuctra	1
Capniidae	
Allocapnia	2
Capnia	1
Perlidae	
Acroneuria	1
Atoperla	1
Neoperla	1
Perlesta	4
placida	4
Perlinella	2
Perlodidae	
Hydroperla	1
Isoperla	2
Chloroperlidae	
Chloroperla	3
MEGALOPTERA	
Sialidae	
Sialis	4
Corydalidae	
Chauliodes	4
Corydalus	3
Nigronia	2
NEUROPTERA	
Sisyridae	1
TRICHOPTERA	
Hydropsychidae	
Cheumatopsyche	6
Diplectrona	2
Hydropsyche	5
arinale	5
betteni	5
bidens	5
cuanis	5
frisoni	5
orris	4
phalerata	2
placoda	4
simulans	5
Macronema	2
Potamyia	4
Symphitopsyche	4
Philopotamidae	
Chimarra	3
Dolophilodes	0
Polycentropodidae	
Cyrnellus	5
Neureclipsis	3
Nyctiophylax	1
Polycentropus	3
Psychomyiidae	
Psychomyia	2
Glossosomatidae	
Agapetus	2
Protoptila	1

MACROINVERTEBRATE	TOLERANCE VALUE
Hydroptilidae	
Agraylea	2
Hydroptila	2
Ithytrichia	1
Leucotrichia	3
Mayatrichia	1
Neotrichia	4
Ochrotrichia	4
Orthotrichia	1
Oxyethira	2
Rhyacophilidae	
Rhyacophila	1
Brachycentridae	
Brachycentrus	1
Lepidostomatidae	
Lepidostoma	3
Limnephilidae	
Hydatophylax	2
Limnephilus	3
Neophylax	3
Platycentropus	3
Pycnopsyche	3
Phryganeidae	
Agrypnia	3
Banksiola	2
Phryganea	3
Ptilostomis	3
Helicopsychidae	
Helicopsyche	2
Leptoceridae	
Ceraclea	3
Leptocerus	3
Mystacides	2
Nectopsyche	3
Oecetis	5
Triaenodes	3
COLEOPTERA	
Gyrinidae (larvae only)	
Dineutus	4
Gyrinus	4
Psephenidae (larvae only)	4
Psephenus	4
herricki	4
Eubriidae	4
Ectopria	4
thoracica	4
Dryopidae	4
Helichus	4
lithophilus	4
Helodidae (larvae only)	7
Elmidae	
Ancyronyx	2
variegatus	2
Dubiraphia	5
bivittata	2
quadrinotata	7
vittata	7
Macronychus	2
glabratus	2
Microcylloepus	2
Optioservus	4
ovalis	4
Stenelmis	7
crenata	7
vittipennis	6

MACROINVERTEBRATE	TOLERANCE VALUE
DIPTERA	
Blephariceridae	0
Tipulidae	4
Antocha	5
Dicranota	4
Eriocera	7
Helius	5
Hesperoconopa	2
Hexatoma	4
Limnophila	4
Limonia	3
Liriope	7
Pedicia	4
Pilaria	4
Polymeda	2
Pseudolimnophila	2
Tipula	4
Chaoboridae	8
Culicidae	8
Aedes	8
Anopheles	6
Culex	8
Psychodidae	11
Ceratopogonidae	5
Atrichopogon	2
Palpomyia	6
Simuliidae	
Cnephia	4
Prosimulium	2
Simulium	6
clarkei	4
corbis	0
decorum	4
jenningsi	4
luggeri	2
meridionale	1
tuberosum	4
venustum	6
verecundum	6
vittatum	8
Chironomidae	
Tanypodinae	
Ablabesmyia	6
mallochi	6
parajanta	6
peleensis	6
Clinotanypus	6
pinguis	6
Coelotanypus	4
Labrundinia	4
Larsia	6
Macropelopia	7
Natarsia	6
Pentaneura	3
Procladius	8
Psectrotanypus	8
Tanypus	8
Thienemannimyia group	6
Zavrelimyia	8
Diamesinae	
Diamesa	4
Pseudodiamesa	1

MACROINVERTEBRATE	TOLERANCE VALUE
Orthocladiinae	
Cardiocladius	6
Chaetocladius	6
Corynoneura	2
Cricotopus	8
bicinctus	10
trifasciatus	6
Eukiefferiella	4
Hydrobaenus	2
Nanocladius	3
Orthocladius	4
Parametriocnemus	4
Prodiamesa	3
Psectrocladius	5
Rheocricotopus	6
Thienemaniella	2
xena	2
Chironominae	
Chironomus	11
attenuatus	10
riparius	11
Cryptochironomus	8
Cryptotendipes	6
Dicrotendipes	6
modestus	6
neomodestus	6
nervosus	6
Einfeldia	10
Endochironomus	6
Glyptotendipes	10
Harnischia	6
Kiefferulus	7
Microtendipes	6
Parachironomus	8
Paracladopelma	4
Paralauterborniella	6
Paratendipes	3
Phaenopsectra	4
Polypedilum	6
fallax	6
halterale	4
illinoense	5
scalaenum	6
Pseudochironomus	5
Stenochironomus	3
Stictochironomus	5
Tribelos	5
Xenochironomus	4
Tanytarsini	
Cladotanytarsus	7
Micropsectra	4
Rheotanytarsus	6
Tanytarsus	7
Ptychopteridae	8
Tabanidae	7
Chrysops	7
Tabanus	7
Dolichopodidae	5
Empididae	6
Hemerodromia	6

MACROINVERTEBRATE	TOLER VAL
Syrphidae	11
Ephydridae	8
Sciomyzidae	10
Muscidae	8
Athericidae	
Atherix	4
MOLLUSCA	
GASTROPODA	
Viviparidae	
Campeloma	7
Lioplax	7
Viviparus	1
Valvatidae	
Valvata	2
Bulimidae	
Amnicola	4
Pleuroceridae	
Goniobasis	5
Pleurocera	7
Physidae	
Aplexa	7
Physa	9
Lymnaeidae	
Lymnaea	7
Stagnicola	7
Planorbidae	
Gyraulus	6
Helisoma	7
Planorbula	7
Ancylidae	
Ferrissia	7
PELECYPODA	
Unionidae	
Actinonaias	
carinata	1
Alasmidonta	
marginata	1
triangulata	0
Anodonta	3
Carunculina	7
Elliptio	2
Fusconaia	1
Lampsilis	1
Ligumia	1
Margaritifera	1
Micromya	1
Obliquaria	1
Proptera	1
Strophitus	4
Tritogonia	1
Truncilla	1
Utterbackia	1
Sphaeriidae	
Musculium	5
Pisidium	5
Sphaerium	5
Cyrenidae	
Corbicula	4

Appendix B. **IBI SUMMARY TABLE**

STREAM NAME: Beaver Creek STATION CODE: OIB-02
STATION DESCRIPTION:
COUNTY: Clinton T: 3N R: 3W 1/4 SECT: SW27
COLLECTOR(S): IDOC AIBI COMPUTED BY: R.L. Hite
METHOD: RO STREAM ORDER = 5
DATE OF COLLECTION: 6-30-82 DATE OF CALCULATION: 1-17-89
AREA SAMPLED = .183 ACRES UNIT OF EFFORT:
Additional information:

Number of native species = 16 Species metric factor = 3
Number of sucker species = 2 Sucker metric factor = 3
Number of sunfish species = 2 Sunfish metric factor = 3
Number of darter species = 1 Darter metric factor = 1
Number of intolerant species = 1 Intolerant metric factor = 1
Prop.(%) of Green sunfish = 29.78723 Green metric factor = 1
Prop.(%) of Hybrids = 0 Hybrid metric factor = 5
Prop.(%) of omnivores = 6.808511 Omnivore metric factor = 5
Prop.(%) of insect. cypr. = 5.957447 Ins. cypr. metric factor = 1
Prop.(%) of carnivores = 3.829787 Carnivore metric factor = 3
# of fish/0.1 ac = 128.4153 Condition factor = 2.454546
Abundance factor (based on sampling method) = 1
Total abundance = 235
Total number of species = 16

The AIBI for this site is: 29.45455

SPECIES ABUNDANCE TABLE

	Common name	Scientific name	Abundance
1	grass pickerel	Esox americanus	9
2	carp	Cyprinus carpio	4
3	golden shiner	Notemigonus crysoleucas	12
4	red shiner	Notropis lutrensis	1
5	sand shiner	Notropis stramineus	1
6	redfin shiner	Notropis umbratilis	12
7	white sucker	Catostomas commersoni	2
8	bigmouth buffalo	Ictiobus cyprinellus	1
9	black bullhead	Ictalurus melas	3
10	yellow bullhead	Ictalurus natalis	19
11	tadpole madtom	Noturus gyrinus	11
12	pirate perch	Aphredoderus sayanus	76
13	blackstripe topminnow	Fundulus notatus	11
14	green sunfish	Lepomis cyanellus	70
15	longear sunfish	Lepomis megalotis	1
16	slough darter	Etheostoma gracile	2

PERSPECTIVES IN FISH SAMPLING AND ANALYSIS TO MONITOR BIOLOGICAL INTEGRITY OF RECEIVING WATERS

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Abstract

There is a legal mandate as well as an ecological imperative to promote biological monitoring of receiving waters. There are many tools available to us, and certainly the conceptual basis of Karr's IBI model has made an important contribution to water quality assessment. I view the IBI in the context of an evolving process; the IBI is not the focal point, rather it is the community concept upon which "biotic integrity" is based that is of fundamental interest. Thus, from a national perspective it would be unwise to center on a particular, single index or phylogenetic group to monitor biological integrity; however every assessment should attempt to consider structural, functional and population characteristics (Karr 1981) which reflect water quality or habitat alteration. For several reasons, I continue to prefer to use fish to monitor water quality. However, their usage implies several constraints which I have emphasized. Precautions must be taken to maximize representative sampling. Cairns' (1977) views are appropriate: "It is evident that no single method will adequately assess biological integrity nor will any fixed array of methods be equally adequate for the diverse array of water ecosystems. The quantification of biological integrity requires a mix of assessment methods suited for a specific site and problem . . . What is needed is a protocol indicating the way in which one should determine the mix of methods that should be used to estimate and monitor threats to biological integrity."

The Legal Mandate

The environmental impact assessment (EIA) procedure and the accompanying environmental impact statement (EIS) process were legally mandated in the National Environmental Policy Act (NEPA) of 1970. NEPA was a procedural reform to institutionalize environmental considerations into the Federal planning and decision-making process (Dickson et al. 1975). The basic intent of NEPA was to require that environmental considerations be evaluated in relation to social, economic and technological factors in policy, program and project determinations. Specifically, it was required that all Federal agencies prepare a detailed EIS for actions that may affect environmental quality; environmental impacts, mitigating measures and alternatives must be considered in the EIS (Burton et al. 1983). A basic assumption of NEPA was that procedures (EIS) which generate better information will result in better decisions, however this is not guaranteed. For instance, NEPA did not prohibit authorization of projects which have adverse impact, rather it was concerned with the procedural documentation of these impacts (Fairfax & Burton 1983).

Most states have enacted similar EIS/EIA requirements in recent years. Additionally, a suite of Federal legislation was passed which strengthened NEPA in concept.

Perhaps foremost among these from an aquatic ecosystem perspective was the Federal Water Pollution Control Act of 1972 (PL92-500) which created the U.S. Environmental Protection Agency and set effluent limitations on industrial point-source discharges based on availability and economics of control technology (Hocutt 1981). The stated intention of PL92-500 was to " . . . restore and maintain the chemical, physical and biological integrity of the Nation's waters."

Other legislation impinging on the aquatic environment included the Clean Water Act, Toxic Substances Control Act and Ocean Dumping Act, among others. The Clean Water Act of 1977 amended the Federal Water Pollution Control Act of 1972 and broadened the regulations to monitor and improve water quality. The Clean Water Act defined pollution as ". . . the manmade or man-induced alteration of the chemical, physical, biological, and radiological integrity of water." Equally important to the NEPA spirit, but with a perspective of expanding public involvement, was the Freedom of Information Act of 1974 which assured public access to all public records except those falling under restricted classifications and granted citizens the right to sue those federal agencies which wrongly withhold information. In this same vein, the Federal Advisory Committee Act of 1976 and the Government in the Sunshine Act sought to increase public involvement in the decision making process, ultimately requiring that proposed Federal actions be publicly announced in the Federal Register (Fairfax & Burton 1983).

Section 304(a) of The Water Quality Act of 1987, the most recent amendment of PL92-500, has focused on the development of biological criteria and the use of instream biological data to monitor water quality. Section 304(a)(8) directs "The Administrator, after consultation with appropriate State agencies and within 2 years after the date of the enactment of The Water Quality Act of 1987, shall develop and publish information on methods for establishing and measuring water quality criteria for toxic pollutants on other basis than pollutant- by-pollutant criteria, including biological monitoring and assessment methods." In effect, the amendment emphasized the broadening of the range of criteria used to ensure compliance of standards set by the NPDES permits, and signifies a shift from pipe standards philosophy to receiving system impact.

Environmental Stress

Stress in the aquatic environment is usually viewed as man-related. However, stress may also be a natural phenomenon (Hocutt 1985); examples are (1) elevated seasonal temperatures with a corresponding decreased in saturated oxygen levels, (2) shifting substrates, and (3) fluctuations in salinity regimes. Stress can act on aquatic organisms either directly through toxic modes or indirectly through alterations in the food chain or reproductive behavior, for example. Also, stress can be viewed as being selective or non-selective in its nature. If selective, the elimination of target species with low thresholds may be observed, however this could be accompanied by increased productivity of surviving taxa. If the stress is non-selective, species richness may not decrease although overall biomass would be expected to decline.

Stress levels are usually determined by the intensity (i.e., concentration), nature (e.g., the half-life or bio-degradableness of a pollutant), mode (e.g., temperature, pH, heavy metal, pesticide, etc.), duration and rate of exposure of the organism or community to the stress. From a biological perspective, stress will be dependent upon the species, its life stage and sex, and the presence of other flora and fauna. A low-intensity stress may result in little damage even over a prolonged period of time, however, if the stress is increased either by intensity, rate of exposure or the introduction of a synergist, the probability of ecosystem damage is increased.

It is recognized that physical, chemical, radiological and biological perturbations can have a deleterious, sometimes irreversible, impact on the structure and function of impacted systems. However, it is also recognized that the environment can be used as an extension of the water treatment facility if the assimilative capacity of the system is not exceeded (Cairns 1977). Thus, environmental assessment can be viewed by two central themes: (1) water resources management, and (2) water quality assessment in terms of stress and recovery of a damaged ecosystem. It is always preferable, however, to operate within the limitations of the former to avoid the latter.

Environmental Measurement of Biological Integrity

Historically, physicochemical parameters have been given precedence over biology in the study of stressed aquatic ecosystems. Chemical evaluation of stressed conditions allows identification of the substances involved and their concentrations. This fact is central to the National Pollution Discharge Elimination System (NPDES), and its enforcement by the USEPA. However, such measurements are ineffective in estimating the synergistic affects of multiple effluents on aquatic biota, or long-term sublethal effects. Additionally, physicochemical measurements may well miss the short-term, highly concentrated discharge critical to assessment of biological impact, or other man-induced physical alterations of the environment (Karr 1981). As such ". . . pollution is essentially a biological phenomenon in that its primary effect is on living things" (Hynes 1971). Mackenthum (1969) and Hynes (1971) outlined the history of aquatic biology and its relationship to pollution effects.

Biomonitoring for NPDES compliance requirements has centered on the use of bioassay procedures rather than biosurvey methodology. Biosurveys are reported (e.g. Roop & Hunsaker 1985) to be too expensive and time consuming to warrant consideration for rapid site specific assessments; however, these arguments are weak in comparison to the fact that aquatic communities in situ are integraters of past and present environmental conditions. As well, bioassay procedures have several restrictions in their use as a holistic approach to environmental assessment: (1) laboratory-based toxicity studies may not adequately reflect ecosystem impact of point and non-point sources of discharge; (2) multiple point sources can act antagonistically or synergistically in the ecosystem; (3) there can be a large inherent variability in the toxicity tests themselves; (4) effluents have high variability, hence mean NPDES standards may not be sufficiently protective; and (5) preferred bioassay test organisms are often chosen for their tolerance to laboratory conditions, and are not necessarily the most sensitive species or life stage.

The quantification, description and comparison of terrestrial plant communities preceded similar advances for aquatic communities. Many of the biosurvey techniques used to assess aquatic ecosystems evolved from Kolkwitz and Marsson's (1908, 1909) saprobien system and Margalef's (1951) diversity index based on information theory, and resulted from the need to assess the effects of pollution. More recently James Karr and his associates have attempted (Karr 1981; Karr and Dudley 1978, 1981) to develop an index of biological integrity (IBI) using fish communities to measure stream degradation. Karr's objectives were not all together different than those of many ecologists [e.g. Cairns and Dickson (1977); Stauffer and Hocutt (1980)], i.e., to develop a system which would have predictive value for determining the amount of stress a system could assimilate, and the potential of a system to recover once it was stressed. Indeed, Karr's work (and that of others) adds emphasis to the pioneer aquatic ecology investigations of Ruth Patrick (1949), W. Beck (1954, 1955) and John Cairns (e.g. 1974) in the United States, who stressed the importance of community assemblages in data interpretation.

The emphasis of ecologists to measure "biological integrity" has been a direct consequence of the Federal Water Pollution Control Act of 1972 (PL92-500), the stated intention (to repeat from above) was to ". . . restore and maintain the chemical, physical and biological integrity of the Nation's waters." Frey (1975) defined biological integrity as "the capability of supporting and maintaining a balanced, integrative, adaptive community of organisms having a species composition similar to that of the natural habitat of the region."

I (Hocutt 1981; Hocutt and Stauffer 1980), like Karr (1981), contend that fish communities should be given preference when assessing man-related impacts in freshwaters. The most compelling reason is that structurally and functionally diverse fish communities directly and indirectly reflect water quality conditions at a given locality in that their community stability is indicative of past and present environmental perturbations (Hocutt 1981). The value of fishes in environmental assessment of estuarine and marine systems is more limited when one takes into account the large-scale migrations of many species, however, fish continue to have great utility when their seasonality of occurrence is considered in relation to their life history aspects. Stauffer and Hocutt (1980) summarized the value of using fish data in assessment of ecological integrity, noting that (1) fishes occupy the upper trophic level in most aquatic systems, and as such, the "healthiness" of the fish community implies the "healthiness" of lower trophic levels and phyletic groups, (2) in their development from larvae to mature adults, fishes pass from the primary consumer stages to subsequently higher levels, (3) fish are relatively easy to identify, thus the use of fish data is made more readily available, and (4) more is generally known for the life histories of fishes than other phyletic groups, thus it is easier to relate structural and functional relationships in fish community assemblages.

There are, however, some restrictions to the use of fisheries data for instream biomonitoring. Karr et al. (1986) identified four problem areas in sampling stream fishes accurately for an IBI analysis: (1) Purpose of data gathering must be IBI oriented to obtain a representative sample; (2) sampling gear, water conditions and fish behavior can affect accuracy; (3) the range of habitats sampled has a major effect; and (4) atypical samples result when unrepresentative habitats (e.g., beneath bridges) are next to the sample site. Additionally, I have emphasized the qualitativeness of fish collecting (Hocutt 1981), and the fact that fish may at times be totally unsuitable for monitoring ecological integrity. For instance, fish data may not accurately reflect (1) the biological purity of the water, (2) the occurrence of tastes or odors, (3) substances physically or chemically harmful to other life forms, (4) the suitability of our water source for specific industrial requirements, or (5) the desirable use of a water body for human consumption (Brown 1978).

The "advantages" of the IBI can be debated, however it remains a fact that the single most important parameter of the conceptual design of the IBI is its reliance on the structural and functional properties of the (fish community). The advantages of the IBI are reported to be: (1) It is quantitative and provides criteria to determine what is excellent or poor; (2) It uses several attributes to reflect conditions - no single attribute can reliably indicate degradation but the IBI is correlated with degradation; (3) There is no loss of information in calculating the index value -- the metric values are available to pinpoint the ecological attributes that are being altered; and, (4) Professional judgment is applied in a systematically and ecologically sound manner - this occurs when establishing metric scoring criteria, not when interpreting the index value as with most assessment methods (Miller et al. 1988). Due to the flexibility of the IBI model to be modified, it has been adapted for regulatory use in Ohio and Illinois and is currently being considered for formal adoption at the national level as a means of monitoring water quality (Miller et al. 1988).

It must be stated, however, that professional judgment remains a key issue from the moment of study design, through the field phase and especially in data interpretation. Every professional is a product of their schooling and experience; thus, while professional judgment can be a strength, it most certainly may be a weakness - and if not a weakness then a valid contrast in opinion. For example, Leonard & Orth (1986) used a modified six-metric IBI for Appalachian streams, but Angermeier & Karr (1986) included all 12 original metrics in their interpretation of the same data.

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ECOREGIONS: AN APPROACH TO SURFACE WATER PROTECTION

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Abstract

Many of our most important scientific and management questions require some sort of regionalization. Problems are too widespread and numerous to be treated on a site by site basis and ecosystems are too variable to be treated the same way nationwide. This paper demonstrates the use of a regional framework for determining chemical and biological goals for surface waters. In four case studies, an ecoregion map drawn from landscape characteristics was used to stratify the naturally occurring variance in water quality and biological communities. An ecoregion framework helps us apply sound ecological theory to setting goals for entire states or regions of the country. Such a framework is an important bridge between site-specific and national approaches. When combined with appropriate statistical design, the ecoregional approach can provide precise expectations about large numbers of water bodies that would not be possible from traditional site-specific research or river basin surveys.

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